

**‘The seasonal influence of large aquatic plants at contrasting study sites
on the River Frome, Dorset’**

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by Kelly Joanne Watson, BSc

**Thesis submitted to the University of Nottingham for
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ABSTRACT

This thesis utilised the River Frome, Dorset, as a study catchment within which to examine the seasonal effects of large aquatic plants, or ‘macrophytes’, on hydraulic roughness, water velocity, river stage and fine sediment distribution. The thesis formed part of LOCAR, a NERC thematic research programme, and was motivated by the need to provide empirical data to improve river management and to help achieve a compromise between vegetation management for flood control and for maintaining and maximising biodiversity.

The thesis employed a hierarchical research design with linked data collection at macro-, meso-, and microscale. (i) Macroscale research employed archival River Habitat Survey data and primary river surveys to place the more detailed meso- and microscale work in context. The analyses revealed the existence of a distinct chalk river group and examined the place of the River Frome within this. This provided contextual information to aid extrapolation of the current findings and facilitates comparisons with previous and future research. (ii) Mesoscale research focused upon stage and discharge measurements, and grid-based measurements of hydraulic variables. The analyses showed that macrophytes can have demonstrable and quantifiable effects on hydraulic roughness and sediment storage, which causes seasonal change in the stage/discharge relationship. However, this effect varies according to channel morphology and riparian land use and is subject to the attainment of a critical biomass. (iii) Microscale analyses employed high-frequency turbidity probes to investigate sediment processes within macrophyte beds. Each vegetation patch acted as a unique sediment filter, the characteristics of which changed over the growing season, and varied with distance along the patch. The results also suggested that retention of fine sediment is size selective and varies according to plant architecture and in-channel location.

The thesis findings imply that vegetation management must be approached with greater sensitivity to reach scale and sub-reach characteristics.

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This thesis relies heavily on empirical data, collected in ungainly chest waders on freezing January mornings and baking July days, and I am grateful to all who suffered with me. Special thanks go to Gemma Harvey, who took time out of her own PhD to help me complete my fieldwork. I hope I have managed to repay at least some of my debt at the River Tern. I would also like to thank Dr Joanne Goodson (then at Kings College London) and Helen Moggridge for their much needed help and support and sense of humour on field work trips. I am also extremely grateful to my parents for twice travelling from Ireland to Dorset to help me. Many thanks also to Ian Conway, Teresa Needham and Graham Morris for their essential help with fieldwork kit and laboratory work. I would also like to thank a succession of office mates and housemates in Nottingham for vital company in the often lonely world of a PhD student.

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1. INTRODUCTION

1.1 Research rationale

This thesis examines the seasonal effects of large aquatic plants, or ‘macrophytes’, on hydraulic roughness, water velocity, river depth and the distribution of fine sediment at contrasting study sites on the River Frome, Dorset. The hydraulic effects of macrophytic plants have been the subject of research for many decades, but this research has evolved in a disparate and speculative manner and is full of theory and contradiction, while at the same time is lacking in empirical data or quantified results. This thesis aims to consolidate published knowledge and to provide new data and analyses to examine the contradictions and conflicts apparent in previous research. It also seeks to provide quantified estimates of the hydraulic and sediment impacts of large aquatic plants. The thesis is motivated by the need to provide empirical data to help inform river managers and to help attain a compromise between vegetation management for flood control and for maintaining and maximising biodiversity. Two central themes run through the thesis, encompassing seasonal changes in velocity, sediments and river stage: i) whether macrophytes can influence river stage through reach scale velocity changes, and whether this effect is subject to a critical biomass, and ii) whether macrophyte induced changes in fine sediment storage can affect river stage.

The thesis was undertaken as part of LOCAR (Lowland Catchment Research): a NERC (Natural Environment Research Council) thematic research programme focusing on the ‘hydro-environment’ of permeable lowland catchments in England.

LOCAR focused on the river catchment as a research unit, but instead of one project spread thinly throughout a single catchment, it involved 12 major projects, across three contrasting catchments, each of which considered different aspects of hydrology and ecology. Overall, the LOCAR programme aimed to “improve the science required to support current and future management needs for permeable lowland catchments through an integrated and multi-disciplinary experimental and modelling programme” operating at “different spatial and temporal scales” and considering “different land uses”

(Wheater *et al.*, 2004 and <http://www.nerc.ac.uk/research/programmes/locar/aims>).

1.2 LOCAR: Lowland Catchment Research

LOCAR was established in 2000, and ran until 2006, with the aim of creating high quality UK field research facilities. The intention is to use these facilities to promote detailed, interdisciplinary, research relating to the input-storage-discharge cycle and stream and wetland habitats of lowland groundwater dominated river systems.

The main LOCAR Science Questions were:

1. *What are the key hydrological processes controlling surface water-groundwater interactions, the movement of groundwater, and material fluxes in lowland permeable catchments?*
2. *What are the key physical, chemical and biological processes operating within the valley floor corridor which affect the surface water and groundwater?*

3. *How do varying flow regimes control in-stream, riparian and wetland habitats?*
4. *How does land use management impact on lowland catchment hydrology, including both water quantity and quality, and wetland ecology?*
5. *How can the hydrological, hydrogeological, geomorphological and ecological interactions resulting from natural or anthropogenic changes be predicted using integrated mathematical models?*

(<http://www.nerc.ac.uk/research/programmes/locar/aims>).

This thesis is centred within LOCAR Science Questions 1, 2 and 4 and considers: storage of in-channel sediment; biological processes which affect surface water and the impact of land use management on catchment hydrology and wetland ecology. The thesis also has relevance for Question 3, but runs slightly counter to it: the thesis examines how in-stream vegetation can control physical river variables such as average velocity and can counteract seasonal changes in river stage.

1.2.1 The LOCAR catchments and the River Frome

The LOCAR programme centred on three instrumented lowland permeable catchments in England. Two are in chalk catchments: the Frome/Piddle in Dorset and the Pang/Lambourn in Berkshire, and the other is a sandstone river: the River Tern in Shropshire. Within LOCAR, this PhD project operated as part of a larger

group endeavour entitled: ‘Vegetation influences on fine sediment and propagule dynamics in groundwater-fed rivers: implications for river management, restoration and riparian biodiversity’. The group aim was to integrate observations of river flows, fine sediment, vegetation biomass, channel morphology and seed and propagule transfers to develop a classification of groundwater river reaches. All datasets generated through LOCAR projects are to be placed in the United Kingdom Environmental Data Index (UKEDI) and will be accessible to interested parties who may wish to further LOCAR research (<http://ukedi.ceh.ac.uk/>).

Lowland rivers were chosen for investigation under LOCAR because they are regarded as under-researched in the United Kingdom. Lowland rivers are also arguably the most pressured part of the river system: they flow through the most heavily populated areas and suffer most from the effects of agriculture; urbanisation; flood defence measures and water abstraction (Wheater *et al.* 2005). An improved scientific base, developed through LOCAR, may help mediate these effects and improve river management. Similarly, groundwater rivers are relatively understudied and are particularly under pressure in lowland areas because of groundwater abstraction. Groundwater rivers are also especially useful in the context of this study; they are generally thought to have a more regulated and less flashy river regime than impermeable catchments and this makes it easier to isolate seasonal changes imposed by vegetation cycles. The larger group project encompasses two of the five LOCAR rivers: the River Tern in Shropshire and the River Frome in Dorset, while this PhD project concentrates solely on the River Frome.

The River Frome was chosen from the three prospective LOCAR catchments because it is the southernmost chalk river. Both the southern location and the chalk bedrock contribute towards a higher plant biomass and a more marked seasonal variation in macrophytic vegetation cover; the southern location suggests a warmer climate and a longer growing season, while the chalk rock ensures a high availability of plant nutrients. This stronger vegetation signal means that vegetation effects may be more easily isolated from other factors. The choice of a high biomass river also concentrates research where macrophyte growth is likely to be regarded as problematic and where management may be required.

1.3 The research aims

Several aims and objectives were formulated to elucidate the specific aspects of the LOCAR science questions covered by this thesis. These aims help direct the thesis and will be used to evaluate the effectiveness of the research. The first research aim frames the research and provides a context for the River Frome, the study sites and the analytical results:

1. *To place chalk rivers, and more specifically the River Frome and the chosen study sites, in a national context in terms of physical river characteristics and macrophytic vegetation type and abundance.*

There are also four analytical research aims which investigate the complex interactions between macrophytic vegetation, water velocity, water depth and

sediment. These aims consider macrophyte effects over a range of discharges and over changing seasons and endeavour to:

2. *Examine the influence of macrophytic plants on the stage/discharge relationship.*
3. *Investigate the effects of macrophytes on microscale and reach-scale flow resistance and velocity.*
4. *Summarise the shifting distribution of fine sediments both seasonally and spatially, at the meso- and microscales, in order to assess the impacts of macrophytic vegetation on sediment processes and identify the mechanisms that control sediment retention within macrophyte beds.*
5. *Describe the sediment filtering effect of macrophyte plant beds - to determine whether deposition within macrophyte beds is a size selective process and to define the 'pore size' of the proposed vegetative 'filter'.*

Finally, the thesis is at all times directed towards one all encompassing aim, in line with the main aim of LOCAR, which places the research in a management context:

6. *To contribute to the scientific base needed for the sustainable management of lowland groundwater-fed river systems, via the provision of empirical data to help inform river managers of the optimum macrophytic vegetation cover in streams and of the appropriate timing and methods of management.*

Each aim has several specific objectives associated with it, and these have been listed in Table 1.1. These research aims are explored in later chapters of the thesis and it is hoped that this research will contribute significantly to the understanding of the effects of macrophytes on stage, velocity and sediment in chalk streams. A nested research design was obtained employed to address the research questions and involved linked investigations at three research scales (macro-, meso- and microscale) of decreasing spatial coverage but of increasing complexity. Macroscale investigations involve data collection at spatial ranges greater than 100m and within a timescale ranging from months to years, mesoscale investigations are conducted at spatial scales of 10s to 100s of metres and within a time framework of hours or days and microscale investigations were based at spatial scales of mm or cm and on a time framework of seconds or minutes (Carling, 1995).

It was envisaged that investigations at one scale would help inform investigations at other scales to form a more integrated picture of macrophyte influence and to help apply the results to sites and rivers outside of those studies in the thesis. The amount of data and analysis possible in any thesis, is ultimately limited and the majority of the data collected for the thesis, and the greater part of the analysis, were carried out at the meso-scale, considered to be the scale considered most relevant to river managers (large enough to and small enough to enable tailored management and to focus management where it is most needed). In keeping with the aims of LOCAR (Gash, 2006), the macroscale analyses aimed to provide a catchment-wide perspective for the results and were carried out to aid application of the mesoscale results beyond their immediate site context (at other sites on the River Frome and in other chalk catchments), while the microscale analyses addresses some essential

Table 1.1 The six thesis aims and associated research objectives which drive the thesis. There are four analytical research aims and two framing aims that place the thesis in a national and management context.

Research aim	Key research objectives
<i>1. To place Chalk rivers, and more specifically the River Frome and chosen study sites, in a national context. (investigations undertaken at the macroscale)</i>	<ul style="list-style-type: none"> • What defines a typical 'chalk' river? • How do chalk rivers compare to rivers from different geological and geographical areas? • How does the Frome compare to other chalk rivers? • How do the chosen field sites compare to other sites on the River Frome? Can they be considered representative?
<i>3. To examine the influence of macrophytic plants on the stage/discharge relationship (investigations undertaken at the meso-scale)</i>	<ul style="list-style-type: none"> • Does the stage/discharge change with seasonal changes in plant cover, and, if so, how? • Does stage increase linearly with increasing plant cover? Or must some critical plant cover be reached before effects are evident? • In summer, does stage increase linearly with increasing discharge? Or does the compression of plant morphology and biomass loss cause a change in the stage/discharge relationship at high flow? • How will the vegetation effects change due to differences in site morphology, riparian shading and plant biomass? • Are the effects of macrophytes sufficient to cause a significant increase in the frequency and magnitude of overbank flooding in lowland ground-water fed rivers? • Do seasonal sedimentation patterns have any effect on river stage?
<i>2. To investigate the effects of macrophytes on flow resistance and velocity (investigations undertaken at the meso-scale)</i>	<ul style="list-style-type: none"> • Is velocity reduced inside macrophyte beds and, if so, to what extent? • Is there an increase in velocity outside macrophyte beds and how significant is this increase? • Are increases in velocities outside of plant beds sufficient to compensate for decreases inside plant velocities? Is mean reach velocity unaffected by vegetation influence? • Is any compensatory action subject to a maximum biomass limit, after which increases in velocity outside of plant beds can no longer compensate for decreases within? • Or does the self-limitation of stand size by aquatic macrophytes ensure that the critical biomass limit is not reached and compensatory action always occurs? • Does riparian shading or complex channel morphology help keep aquatic biomass below this critical level?

Table 1.1 (Continued) The six thesis aims and associated research objectives which drive the thesis. There are four analytical research aims and two framing aims that place the thesis in a national and management context.

Research aim	Key research objectives
<i>4. To summarise the shifting distribution of fine sediments and to assess the impact of macrophytic vegetation on sediment processes (investigations undertaken predominantly at the mesoscale)</i>	<ul style="list-style-type: none"> • What are the preferential depositional areas for fine sediment in each season? • What are the changes in sediment retention through the year? Does a seasonal cycle of retention and release exist? Is there a critical plant cover when sedimentation begins? If so, when does this occur? • Is sedimentation increased within macrophyte beds? • What are the mechanisms that bring about increased sedimentation within macrophyte beds? Are erosion rates reduced or are depositional rates increased? Or, do both processes combine to increase sediment retention in macrophyte beds? • At what distance inside a plant stand does deposition begin and end? • Is sediment eroded from preferential 'flow channels' created between vegetation? • Are changes in 'flow channels' a result of higher erosion rates or lower deposition rates? Or, perhaps, a combination of both?
<i>5. Describe the sediment filtering effect of macrophyte plant beds (investigations undertaken at the microscale)</i>	<ul style="list-style-type: none"> • Is sediment retention within macrophyte beds size selective? • What is the 'pore size' of the vegetative filter? What is the range of particle sizes retained by the plant and which sizes are not retained? • What is the capacity of the vegetative filter? Does an upper limit of retention exist? If so, what might this be? What are the changes in suspended sediment characteristics before and after vegetation stands? • Do both suspended sediment and bedload contribute to deposition within macrophytes? • Does the length of a macrophyte stand affect its filter characteristics and capacity?
<i>6. To contribute to the scientific base needed for the sustainable management of lowland groundwater-fed river system (informed by investigations at three scales: macro-, meso- and microscale)</i>	<ul style="list-style-type: none"> • Do macrophytes increase stage levels in summer relative to winter? • Do macrophytes significantly increase the magnitude and frequency of flooding? • Is a critical vegetation biomass required before effects on stage become evident? • What time of the year is this critical biomass attained? • Does riparian shading help keep the biomass below this critical level? • Do differences in channel morphology have an influence on the extent of stage increases? • Would restoration of uniform channels reduce the impact of macrophytes on stage? • When is the most effective time for management intervention? • What alternatives to traditional management may be available?

questions that it was not deemed feasible to investigate at the mesoscale. The microscale investigations introduce experimental methodologies that were deemed to have promise and may take research forward at this scale. The scales of investigation employed for each of the research aims is detailed in Table 1.1 and methods and time-scales of the research design are described in detail in Chapter 2.

The remainder of the chapter provides an introduction to macrophytes and summarises current knowledge of the effects of macrophytes on velocity, water depth and stage. The perceived need for macrophyte management and the differences between traditional and modern management techniques are also explored. Finally, the chapter concludes with an outline of the structure and content of the rest of the thesis.

1.4 Macrophytes: definition and influence

In studying ‘macrophytes’ it was essential that a workable definition of the term was sought. This was obtained by the amalgamation of two existing definitions, those by Westlake (1975) and Jeffries and Mills (1990). Westlake described the term macrophyte as providing a useful one word expression to describe “all green plants, whether floating or submerged or emergent” (Westlake, 1975, p. 107). This definition demonstrates that macrophytes encompass all plant growth forms, while Jeffries and Mills (1990 p. 63) qualify this with a size restriction, preferring “large plants visible to the naked eye”. The term has no real taxonomic meaning and encompasses vegetation types as diverse as angiosperms (flowering plants), gymnosperms (plants that produce naked seeds), pteridophytes (ferns); bryophytes

(mosses) and large algal species e.g. *Chara* (Westlake, 1975; Jeffries and Mills, 1990 and Allan, 1995). These taxonomic groups are known to grow in close proximity and, given their similar size, respond to the same environmental constraints (Jeffries and Mills, 1990). Macrophytes are normally classified according to their growth form and mode of attachment to the sediment and most classifications yield four major groups, which together encompass a continuum of growth forms: (i) ‘emergents’; (ii) ‘floating leaved and rooted in sediment’ (iii) ‘submergent’ and (iv) ‘free-floating’. Table 1.2 displays the classification given by Jeffries and Mills (1990) and provides a description of the four main macrophyte groups. A brief introduction to macrophyte ecology is provided in Appendix A. This explores macrophyte growth and reproduction and the factors which determine vegetation abundance and distribution within the river channel.

Two plants prevalent in the River Frome, and dominant in the study reaches used for fieldwork in this project, are the submergent *Ranunculus calcareous* (water crowfoot) and the emergent *Sparganium erectum* (branched bur reed). A schematic diagram of the differing growth forms of these species is displayed in Figure 1.1, along with a photographic representation of the general location of the plants in the river channel. *Ranunculus calcareous* is a dense plant made up of many short ‘tassel-like’ leaves which branch into 50 – 150 filaments, and which occur at regular intervals along multiple stems (Dawson, 1976 and 1979). The plant stems are submerged and grow upwards from the stream bed to just below the water surface in a downstream direction. The plant stand is generally wider at the upstream end and tapers towards the downstream end to form an obconical shape when viewed from above (Dawson, 1979). *Sparganium erectum* also forms dense communities, which

Table 1.2 Classification of macrophytes according to plant growth form, adapted from Jeffries and Mills (1990).

Group	Subgroup	Common examples from chalk rivers
1. Macrophytes attached to substrate	Emergent	<i>Myosotis Scorpiodes</i> (Forget-me-not) <i>Nasturtium officinale</i> (Water cress) <i>Veronica Beccabunga</i> (Brooklime) <i>Sparganium erectum</i> (Branched bur-reed)
	Floating-leaved	<i>Nymphaea nuphar</i> (Water lilly) <i>Potamogetons natans</i> (floating leaved pondweed)
	Submergent	<i>Ranunculus calcareous</i> (Water crowfoot) <i>Myriophyllum alterniflorum</i> (Alternate flowered milfoil) <i>Sparganium emersum</i> (Branched bur reed)
2. Macrophytes not attached to substrate, i.e. ‘free floating’	N/A	<i>Lemna minor</i> (lesser duck weed)



Figure 1.1 Schematic diagrams of: (i) the submergent *Ranunculus calcareus* (water crowfoot) and (ii) the emergent *Sparganium erectum* (branched bur reed) adapted from Haslam *et al.*, 1975. Both species occupy characteristic locations in the channel: *Ranunculus calcareus* in the centre of the channel and *Sparganium erectum* at the channel margins.

may be many metres in length, but it has much wider, smooth and ‘keeled’ leaves that are triangular in cross-section and may be up to 1.5 in height. *Sparganium erectum* grows upwards from beneath the water surface; its leaves are rigid and erect and are positioned at right angles to water flow.

1.4.1 The effects of macrophytes on velocity, water depth and sediments

Most researchers agree that large plants impede water flow in rivers and cause a decrease in water velocity within their beds. Champion and Tanner (2000), for example, refer to macrophytic plants as forming ‘semi-permeable dams’ which act as physical barriers and reduce flow velocities. The opinions of other authors are summarised in Table 1.3. As noted from Table 1.3, some researchers suggest that this decrease within vegetated areas acts to reduce overall mean velocity in the river channel. If it is accepted that vegetation does have a significant impact on mean reach velocity at high plant cover, then the effects of plants on water depth and flooding must also be considered. At an equal discharge a reduction in the speed of water flow will necessarily lead to an increase in water depth because the same volume of water must continue to pass through the same river cross-section.

Some authors also describe an *increase* in water velocity in un-vegetated areas outside and above vegetation (see Table 1.3 and Plate 1.1). It could perhaps be argued that the increases in velocity outside vegetation beds may compensate for the decreases within, and that, overall, reach velocity and water depth remain relatively unaffected. However, this compensation effect may only operate below a maximum level of plant cover or abundance and it is plausible to suggest that a critical plant

Table 1.3 Methods and opinions from previous research on effects of macrophytic vegetation on water velocity, water depth and flooding. Blank fields in the table indicate that the author has not expressed an opinion on this issue in the text.

Author(s)	Scale of study	Velocity reduction within vegetation	Increased velocity outside/above vegetation	Increase in water depth	Flood risk
Butcher, 1933	Macroscale	Yes		Yes	Yes
Casey and Newton, 1973	Macroscale			Yes	
Champion and Tanner, 2000	Mesoscale	Yes		Yes	No
Cotton <i>et al.</i> 2006	Micro- and mesoscale	Yes	Yes		
Dawson, 1978	Mesoscale	Yes		Yes	
Dawson and Robinson, 1984	Mesoscale	Yes	Yes	Yes	Yes
Dodds and Biggs, 2002	Microscale	Yes	Yes		
Green, 2005	Microscale	Yes	Yes	Yes	Yes
Haslam, 1978	Literature review	Yes	Yes	Yes	Yes
Hearne and Armitage, 1993	Literature review	Yes		Yes	Yes
Losee and Wetzel, 1993	Microscale	Yes			
Machata-Wenniger and Janauer, 1991	Microscale	Yes	Yes		
Madsen and Warncke, 1983	Microscale	Yes			
Marshall and Westlake, 1978	Literature review	Yes		Yes	Yes
Marshall and Westlake, 1990	Microscale	Yes	Yes		
Pitlo and Dawson, 1990	Literature review	Yes	Yes	Yes	Yes
Rodwell, 1995	Literature review				Yes
Sand-Jensen <i>et al.</i> , 1989	Mesoscale	Yes			
Sand-Jensen and Mebus, 1996	Microscale	Yes	Yes		
Sand-Jensen, 1998	Microscale	Yes	Yes		
Sand-Jensen and Pederson, 1999	Microscale	Yes	Yes		
Stephan and Gutknecht, 2002	Microscale	Yes			
Wade, 1999	Literature review	Yes	Yes		
Watson, 1987	Micro- and mesoscale	Yes		Yes	
Westlake, 1975	Literature review	Yes	Yes		
Wharton <i>et al.</i> 2006	Micro- and Mesoscale	Yes	Yes		
Wilcock <i>et al.</i> , 1999	Mesoscale	Yes		Yes	



Plate 1.1 Luxuriant growth of *Ranunculus calcareous* on the River Frome, June 2004. Note the smooth water surface in areas of low flow within vegetation beds, which contrasts with the areas of high velocity between stands denoted by disturbance of the water surface.

cover limit exists, perhaps when vegetated areas outnumber un-vegetated areas, and that when this is achieved compensatory action will no longer be effective. Sand-Jensen and Mebus (1996) complicate this hypothesis by suggesting that a definite upper limit to vegetation cover exists and that the critical cover needed to reduce mean velocity may not be reached. It is uncertain whether the natural maximum limit of vegetation cover will be greater or less than the theorised critical vegetation cover needed to induce flooding. The possible existence of a critical biomass, which must be reached before macrophyte cover will have any effect on river stage, will be one of the central foci of this thesis. In a similar manner to the uncertainty in macrophyte-velocity interactions, the influence of vegetation on sediments is poorly understood and is subject to conflicting hypotheses. It is thought that stands of aquatic vegetation will act as filters for suspended sediment by slowing, trapping and modifying the fine sediments that pass through their canopy and that sediment depth within plant beds will increase within vegetation beds and will decrease in high velocity un-vegetated areas. There remains doubt as to whether increase within plant beds can cause an increase in overall sediment depth in a reach and again, whether effects on mean sediment level are subject to a critical plant cover. Whether or not the seasonal sediment changes induced by macrophyte cover have any effect on river stage will be a central theme in this thesis.

There are also seasonal considerations that complicate the action of vegetation on velocity and water depth. Macrophytic vegetation experiences a seasonal pattern of growth and decline and peak vegetation biomass is generally reached during the summer months when river discharges are generally low. In this situation the increase in stage which may occur through velocity reduction is unlikely to induce

over bank flooding (Champion and Tanner, 2000) and indeed, the increase in stage may be necessary to maintain various ecosystem functions (Allan, 1995). Higher discharges may occur during spring and autumn at moderate vegetation biomass, and, in this case, two mechanisms act to temper the effects of macrophytes on stage: (i) the compaction of plant mass and plant shape and (ii) the loss of plant biomass. It has been suggested that these effects allow mean velocity to increase and reduce the vegetation effect on water depth. Vegetation effects on water depth are thought to be greatest at low flow when flood risk is least.

Many authors have based their assertions of velocity reduction in plant beds on either outdated methods (Marshall and Westlake, 1978; Madsen and Warncke, 1983 and Dawson and Robinson, 1984), or on a limited pool of viable work produced by a small number of researchers (e.g. literature reviews by Pitlo and Dawson 1990; Hearne and Armitage, 1993 and Wade, 1996). In addition, field data at a meso or macroscale are limited, and most quantified studies focus on the microscale (Madsen and Warncke, 1986; Machata-Wenniger and Janauer, 1991; Sand-Jensen and Mebus, 1996; Sand-Jensen, 1998; Sand-Jensen and Pedersen, 1999 and Dodds and Biggs, 2002). Larger scale studies have focused mainly on the effects of vegetation on mean velocity and mean hydraulic roughness and very little detailed work has been carried out at the reach scale, especially in terms of mapping velocity patterns in relation to plant stands. The influence of velocity reductions on water depth can only really be investigated at the meso or macroscale, and previous work on the effects of vegetation on water depth is also limited. Most hypotheses regarding the influence of macrophytic plants on sediment are based on assumptions formed through velocity-centred research, based on the premise that macrophytes encourage

sedimentation by reducing water velocity (Dawson and Robinson, 1984; Marshall and Westlake, 1990 and Sand-Jensen and Mebus, 1996), and quantified results are particularly lacking.

The thesis aims set out in Section 1.3 were derived following a review of pertinent previous research and have the intention of investigating questions highlighted by previous studies and to fill current research gaps.

1.4.2 Ecological functions performed by macrophytes

Research papers relating to macrophytes before 1970 are scarce (Butcher, 1933; Owens, 1962; Westlake, 1967; Edwards, 1969 and Sirjola, 1969), and a lack of scientific knowledge led to negative perceptions as to the influence of macrophytes in rivers. Macrophytes were thought to dramatically increase stage and increase the incidence of flooding in rivers (Butcher, 1933) and little was known of the ecological benefits of the vegetation. In early research papers these views are evident in plant nomenclature; often macrophytes were referred to as 'aquatic weeds' (e.g. Pitlo and Dawson, 1990 and Watson, 1987) or 'water weeds' (Harley, 1990). The negative perceptions of macrophytes were, and still are, also reflected in management regimes. Macrophytic vegetation in chalk rivers is intensely managed for flood control (Holmes, 1999) and on most rivers in Southern England vegetation can be cut many times during the growing season. Recent views of aquatic macrophytes are much more knowledge-based and positive; research has shown that macrophytes are highly important in both maintaining and creating diverse habitat in river systems

and there is also doubt as to whether macrophytes do contribute to overbank flooding.

Sand-Jensen (1998) refers to macrophytes as 'ecosystem engineers', defined by Jones *et al.* as "organisms that directly or indirectly modulate the availability of resources to other species, by causing physical state changes in biotic or abiotic materials" (Jones *et al.*, 1994 p.374). Macrophytes influence their environment and that of other organisms by both biotic and abiotic means, altering both the "physico-chemical environment and the biological structure" of streams (Sand-Jensen *et al.*, 1989 p.30). In abiotic terms, plants represent a physical barrier to river flow and act to divert flow around the plant. Velocities are reduced within the plant stands (Madsen and Warncke, 1983 and Dawson and Robinson, 1984) and compensatory increases in velocity occur above and beside the vegetation (Machata and Janauer, 1991 and Sand-Jensen and Mebus, 1996). Macrophytes act as refugia for fish and invertebrates (Carpenter and Lodge, 1986; Wade, 1996; Holmes, 1999; van Nes, 1999; Sand Jensen *et al.*, 1999; Large and Prach, 1999 and Tsujimoto, 1999) and provide shelter from high velocities (Carpenter and Lodge, 1986) and protection from predation (Scheffer, 1999). As a result macrophytes have higher populations of invertebrates relative to other substrates (Allen, 1995) and can accommodate a dense invertebrate fauna (Sand Jensen *et al.*, 1989). Macrophytes are also thought to help establish beneficial habitats outside of their beds through the creation of faster flowing channels. This increases habitat heterogeneity (Allan, 1995, Holmes, 1999, Large and Prach, 1999, Champion and Tanner, 2000) and results in increased biodiversity in the stream as more niche habitats are created. Macrophytic plants may even provide habitat beneficial for other plants; they provide sites of attachment

for epiphytic algae (Butcher, 1933; Carpenter and Lodge, 1986 and Sand-Jensen *et al.*, 1989) and can also alter conditions to the benefit of other macrophytes (Haslam, 1978) creating an aquatic succession sequence.

Another abiotic benefit of macrophytes is their effect on water quality (Merezhko, 1973; Large and Prach, 1999 and Schulz, 2003). This is especially true of lakes but also of rivers. By acting as a physical obstruction, and by slowing water flow, macrophytes increase sedimentation and reduce the turbidity of lakes and rivers (van Nes, 1999). Macrophyte plants provide an important link between the water and sediments and sediment trapping in macrophytes is important in nutrient cycling (Barko *et al.*, 1991). Macrophyte actions are especially important under eutrophic conditions, as trapped sediments are useful temporary stores of excess phosphorus and nitrogen. The high invertebrate populations of macrophytic plants serve to increase nutrient processing potential (Sand-Jensen, 1998 and Champion and Tanner, 2000), and help to mediate the effects of excessive nutrient run-off by reducing nutrient loading in downstream lakes and coastal zones (Hearne and Armitage, 1993). The same is true for chemical pollutants which are trapped, fixed and degraded more quickly in the presence of macrophytes (Large and Prach, 1999). Macrophytes may also help in the attenuation of sewage pollution as macrophyte secretions can eliminate some harmful bacteria. Finally, the sediments trapped under vegetation in chalk rivers are predominantly of organic origin (Cotton *et al.* 2006 and Wharton *et al.* 2006) and provide a food source for invertebrates.

Plants also play a more direct, biotic, role in altering and maintaining habitat. Most obviously they are important as a source of oxygen for aquatic organisms (Butcher,

1933; Wade, 1996 and Holmes, 1999). This oxygenation includes not just the water but also the soil; many macrophytes release oxygen into the soil for the use of aerobic microbes (Large and Prach, 1999). Macrophytes also provide a food source for invertebrates, fish and birds (van Nes, 1999). Few invertebrate grazers can directly consume macrophytic plants (Allen, 1995) but they are an important source of food upon decay (Gregg and Rose, 1982). Macrophytes also produce continuous secretions of dissolved organic substances (Sand-Jensen *et al.*, 1989; Jeffries and Mills, 1990; Allan, 1995 and Large and Prach, 1999) which are a nutrient source for biota. Allen (1995) describes macrophytes as important food sources in streams, perhaps not in terms of the volume of material provided but in terms of their seasonal importance. The decay of macrophytic material and its entrance as accessible food into the river system occurs when summer periphyton are in decline, and before the late autumn littoral input into the stream (Allen, 1995); macrophytes, therefore, fill a gap in seasonal food input and availability. Even the suspected increase in stage caused by plants, seen as so unacceptable by some river managers, can be thought beneficial from an ecological viewpoint. For example, Allan (1995) describes macrophyte growth and potentially increased stage as beneficial in mediating high summer temperatures; greater water depth increases the attenuation of sunlight. In addition, higher summer water levels may be needed to prevent desiccation of the eggs of invertebrates (Hearne and Armitage, 1993) and to maintain links between the channel and off-river habitats, particularly with regards to seed dispersal in the riparian zone (Wharton, Pers. Comm.). The maintenance of adequate water levels may be especially relevant in lowland rivers, where many rivers have been channelised and over-widened in the past and where water levels may be reduced through abstraction.

1.4.3 Management of macrophytes: traditional and modern

Traditional management of river macrophytes has centred on the eradication of summer vegetation to prevent flooding and to permit navigation (Holmes, 1999). This was achieved through physical and chemical controls, though neither method is permanent and re-vegetation can often be very rapid. Often it was cost that decided which method should be used (Wade, 1996).

Physical control is the dominant method of macrophyte management. Plants are cut either by hand in small shallow rivers or using mechanical cutters mounted on boats in wider, deeper rivers. The plant material is generally removed after cutting to prevent the regeneration of new plant stands from plant parts and to prevent the adverse effects of decaying vegetation, and as cutting only removes the above ground parts of the plants, the river may be dredged to remove plant roots. The removal of the vegetation has significant ecological effects. Invertebrates are removed from the channel along with the plant material and this has a significant effect on their numbers. Recovery of the mobile invertebrate population generally only occurs after the macrophyte community has re-established itself and less mobile species may take significantly longer (Wade 1996). The macrophyte community itself may be affected and species composition may change significantly. The loss of vegetation also impacts further up the food chain; there is a reduction in the food availability for fish and a loss of shelter.

Chemical control has been used to reduce or remove 'problem' macrophytes under controlled circumstances. Under UK legislation there are restrictions on the types of

herbicides used in and beside water bodies and the dosage that may be used (MAFF, 1995; Environment Agency, 2003). However, even when applied according to the guidelines herbicides can have unwanted direct and indirect effects on the river ecosystem. Some herbicides are 'non-specific' and will affect plant and animal species other than the target plant species and, in a river environment, chemicals may drift outside the target area and affect plants and animals downstream. Toxic effects may occur in invertebrates after prolonged or repeated exposure to low levels of herbicide (Murphy and Barrett, 1990) and toxins may accumulate upwards through the food chain to affect higher predators. Other effects related to herbicide use arise because the affected vegetation is not removed from the river. Decaying vegetation in the river may affect the dissolved oxygen availability and pH in the river and may release toxins and excess nutrients (Wade, 1996). Herbicides generally eradicate all the target plants and these may be replaced by opportunistic undesirable species, such as *Lemna minor* (duck weed). There are also problems of plant populations developing herbicide resistance.

Modern views on the ecological functions of macrophytes have led to necessary changes in management perspectives and actions. Alternatives to traditional clearance have been developed and there has been a shift away from wholesale elimination of vegetation by cutting and herbicide use to less "ecologically traumatic" means of vegetation control (Dawson and Robinson, 1984 p.1944). Use has been made, for example, of biological controls to reduce plant biomass in a less dramatic and more species-specific way. Variations include the use of insects (Harley and Forno, 1990), fungi (Charudattan, 1990) and phytophagous fish (van der Zwerde, 1990). Shading by floating plants (Pitlo, 1978) and by riparian trees

(Dawson and Kern-Hansen, 1979 and Champion and Tanner, 2000) have also been suggested, based on the premise that an increase in riparian shading will limit plant growth in the river channel. Riparian planting represents a return to natural river conditions before tree clearance (Holmes, 1999) and has the attendant benefits of increased allochthonous input, bank stabilisation, and habitat provision. Riparian planting may also be preferable in financial terms if it negates the need for annual management. However, the degree to which changing attitudes in academic and management research have translated into changes in practical management is debateable and flood control is still the overriding priority for many river managers.

New river management practices must ensure a compromise between all user groups, including landowners, recreationalists and conservationists. van Nes *et al.* (1999 and 2002), have attempted an environmental economics approach to macrophyte management in lakes. Using simulation models they calculated that an intermediate biomass would provide optimum benefit to both recreational users and conservationists, but that this was possible only in purely abstract terms. In reality, the maintenance of intermediate biomass would fall short of the true needs of both groups as well as proving prohibitively expensive to implement (van Nes *et al.*, 1999 and van Nes *et al.*, 2003). They suggest the partitioning or zoning of lakes for different purposes with management practices and biomass altered according to the needs of different groups. Champion and Tanner (2000) advocate a similar management strategy for rivers. They suggest alternating open sections of moderate plant biomass and more shaded areas of minimal plant growth to enhance the 'health' of degraded lowland streams.

A recent example of good practice is provided by the River Avon candidate Special Area of Conservation (cSAC) conservation strategy (Wheeldon, 2003). The River Avon is a Site of Special Scientific Interest (SSSI) and a Natura 2000 site, designated partly on the basis of priority macrophytic habitat, and the conservation strategy sets out a weed cutting code of practice (JNCC) which aims to manage the vegetation in such a way as to minimise damage to the conservation interest. Central to the code are the 'criteria for cutting' which describe the circumstances under which cutting of vegetation will represent an overall benefit to the SAC, including when topsoil is saturated; when the rate of rise in the river is likely to lead to the saturation of topsoil within two weeks; when severe poaching occurs due to high groundwater levels (provided stocking levels are appropriate) and when cutting is required to permit compliance with SSSI management agreements. In addition, 'criteria not to be used for cutting' are also listed and the need for cultivation or silage machinery access, for example, will not justify a cut. The code also suggests that, as a guide, a minimum of 25% of *Ranunculus* cover should be retained in each 100m stretch of river and that cutting patterns be varied annually to prevent changes in channel morphology and plant growth. The code also suggests that cut weed should be temporarily deposited on the river bank before being taken off site, to allow invertebrates to return to the river.

A more relaxed management of aquatic macrophytes may actually help to lessen the effect of vegetation on stage. Dawson (1978) suggested that strict annual cutting in some streams may have led to an increase in roughness relative to natural conditions. He suggested that removal of vegetation each summer had led to a more evenly distributed plant rooting area than under unmanaged conditions. In addition, certain

methods of plant harvesting may have led to increased macrophyte biomass by allowing perennating organs to escape downstream and colonise new areas (Champion and Tanner, 2000). Dawson theorised that a return to less systematic cutting and regrowth may lead to a decline in the maximum roughness coefficient. Finally, there may be scope in the future for macrophytes to provide significant societal benefits. For example, it may be possible to manipulate plant density to allow increased abstraction from rivers and from aquifers (Hearne and Armitage, 1993) without causing damaging drops in water level. Plants may also be used to alter flow patterns within streams and could help to mediate the effects of bank erosion, pollution and eutrophication (Sand-Jensen, 1998; Large and Prach, 1999, Champion and Tanner, 2000 and Gurnell *et al.* 2006). Other possibilities include use as a food source for animals and humans, paper and fibre production, wastewater treatment, and biogas production (Joyce, 1990). This thesis aims to provide improved information for river managers as to the hydraulic and sediment effects of macrophytes in rivers, so that flooding concerns may be quantified and may be balanced against the ecological importance of the plants.

1.5 Outline of the thesis

The thesis questions outlined in Section 1.3 of this chapter are explored in detail in the following six chapters. Chapter 2 introduces the River Frome and the selected study sites and sets out the research scales and methods employed in the project. The reasoning behind the selection of sites on the Frome, and the data collection at each, is outlined along with background information on the catchment geology, soils, and land use and the channel geomorphology, river regime, river management and type

and abundance of macrophytic vegetation. In addition, Chapter 2 describes the nested research design employed during the research which involved investigations at the macro, meso and microscale.

Four analytical results chapters make up the main body of the thesis and these are presented in Chapters 3 – 6. Each chapter contains a review of pertinent relevant literature, concepts and ideas, and includes a description of the analytical methods used, a presentation of the collected data and a discussion of the findings. Chapter 3 is based at the macroscale and aims to place the Frome catchment and study sites in context in terms of physical channel characteristics and vegetation abundance. This was achieved through detailed examination of the literature pertaining to chalk rivers (Chapter 2) and through analysis of the Environment Agency's River Habitat Survey archival dataset (RHS Version 3.3, 1994 – 2002). Chapter 4 examines seasonal changes in reach-scale hydraulic parameters, including: river stage; water depth; water surface slope; hydraulic roughness; mean reach velocity and reach-scale sedimentation. Chapter 5 describes results obtained at the sub-reach, grid-based, scale and explores the effects of macrophytes on microscale and reach-scale velocity patterns and the distribution of fine sediments. Chapter 6 presents results obtained during experimental microscale investigations and examines the nature of sediment processes operating in and around individual macrophytes stands and relates these processes to individual plant characteristics.

Chapter 7 provides a conclusion to the thesis and attempts a synergy of all results and findings. Research findings are considered within the framework of the original research aims and the three scales of investigation are drawn together to provide

holistic conclusions. Possible management implications and applications of the research are suggested and the chapter also considers profitable lines of future enquiry.

2. RESEARCH DESIGN, SITE SELECTION AND METHODS

2.1 Chapter synopsis

This chapter has two main sections: (i) a site description section which describes the study catchment, outlines the rationale for site selection on the River Frome and provides a comprehensive description of each study site; and (ii) a methods section which outlines the methods and research design employed in the study. The two sections are necessarily connected in that site selection helped determine which measurement methods were most appropriate and, equally, the data requirements and methods of the thesis helped in selecting suitable study sites. The site description section aims to describe the River Frome catchment, using both primary and secondary data sources, and describes the rationale behind field site selection. A short description of the River Frome based on literature sources and primary field survey data is given to provide a detailed picture of the River Frome and its catchment. The underlying principles behind site selection are outlined and detailed qualitative descriptions of the chosen field sites are given, based on field survey and on personal observations. This includes observations on channel morphology, land use, bank materials, substrate type and riparian and in-channel vegetation. The fixed infrastructure installed at each site, and the range of data collected, are also described.

The methods section outlines the integrated research design and the suite of research methods selected to meet the data demands imposed by the thesis aims set out in Chapter 1. The research design employs a nested hierarchy of investigation

involving linked work at the macro-, meso-, and microscales, as outlined in Table 2.1. This table is presented at the beginning of the chapter to help place the site descriptions in context.

2.2 The catchment in context

To help extrapolate findings from this intensive, single-river study it was necessary to explore the typicality of the Frome as a chalk river and to place the Frome within a wider, national context. To achieve this, the literature relating to chalk streams and the River Frome was explored to discern what typifies a 'chalk stream' and how the Frome complies with these criteria. The findings from the literature-based comparisons are later complemented by more quantitative analyses in Chapter 3.

In England, Cretaceous chalk outcrops over some 21,500 km² (Bradford, 2002), but this is confined to a broad belt that sweeps east from Dorset through to Kent and as far north as Humberside (Bowes, 2004), see Figure 2.1. There are 35 major chalk rivers and tributaries in the UK, more than in any other country (UK BAP Steering Group for Chalk Rivers, 2003). The Dorset Frome is the westernmost English chalk river. It rises from springs near Evershot in the North Dorset Downs and flows through an elongated rural catchment until it reaches the sea at Wareham where it discharges into Poole Harbour. Other rocks are present within the Frome catchment (Figure 2.2), notably Cretaceous Greensand and fluvial sands and gravels, but chalk is the dominant rock type and it outcrops centrally in the catchment, occupying over 46% of the 464 km² total area (Paolillo, 1969 and Environment Agency, 2005).

Table 2.1 Scales of investigation considered in the thesis and the infrastructure and data collection at each of the detailed study sites: Crockways; Hydeclift Plantation; and Frampton Estate.

Scale of investigation	Measurement techniques and analyses	Spatial scope
Macroscale	<p>National River Habitat Survey (RHS) archival database – available from Environment Agency</p> <p>Primary river reconnaissance survey and digital photo survey.</p> <p>Primary RHS surveys in Spring, Summer, Autumn and Winter.</p>	<p>Data from river sites across the UK, describing the character and quality of river sites based on their physical structure.</p> <p>River Frome and selected tributaries (May 2005)</p> <p>Three detailed 500 m study sites on the River Frome: Crockways; Hydeclift Plantation and Frampton Estate (four surveys of each site – April, July and October 2004 and January 2005).</p>
Mesoscale	<p>Time-linked measurement of stage and discharge. Stage measured continuously by three Pressure Transducers (PTs) at each site and discharge measurements taken at one fixed cross-section.</p> <p>Detailed grid-based measurements of: vegetation cover; vegetation growth form; sediment depth; water velocity; and water depth.</p>	<p>Within limits of the 500m RHS reaches at Crockways and Frampton Estate. 49 discharge measurements taken at each site October 2003 – July 2005)</p> <p>20 × 10 m grid-reaches located within the 500 m RHS reaches at Crockways and Hydeclift Plantation (nine surveys of each site January 2004 – January 2005).</p>
Microscale	<p>Collection of vegetation and sediment samples.</p> <p>High-periodicity turbidity probe measurement of the passage of suspended sediment pulses through patches of vegetation.</p>	<p>Crockways grid-reach, October 2003.</p> <p>Crockways and Hydeclift grid-reaches, July 2005.</p>

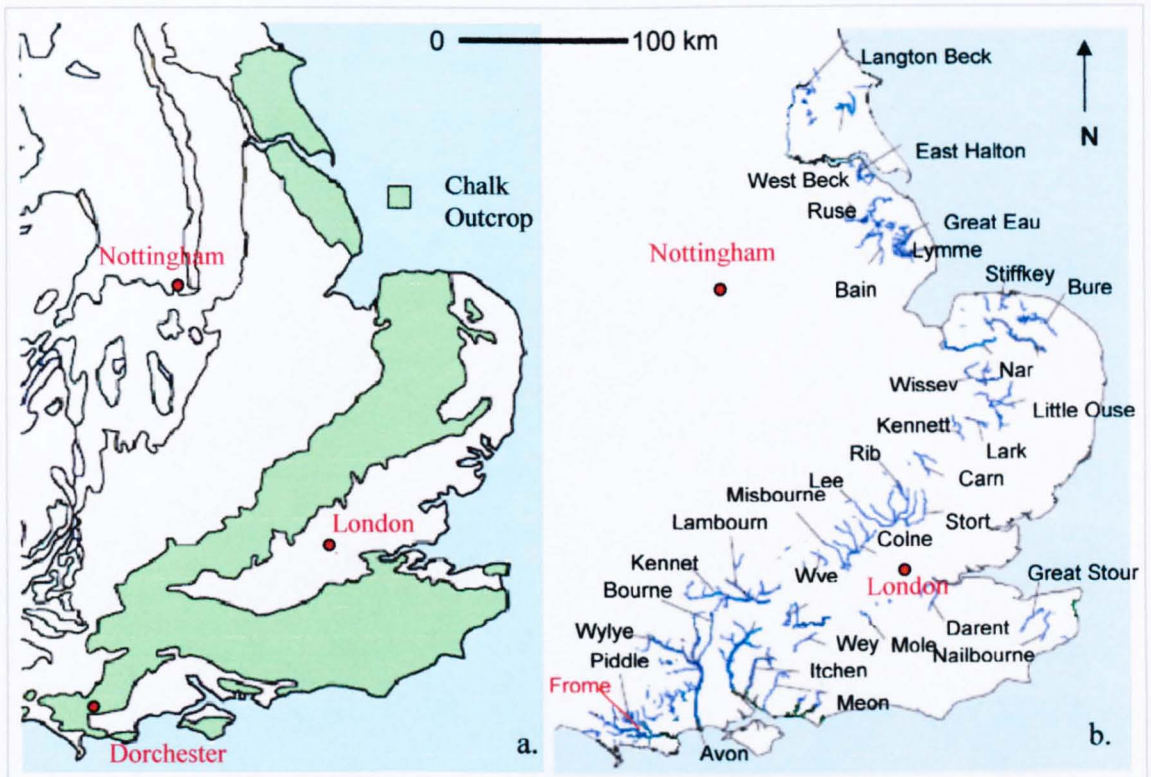


Figure 2.1 Maps showing: a) the location of the chalk outcrop in England; and b) the distribution of the main chalk rivers. Adapted from UK BAP Steering Group for Chalk Rivers (2004).

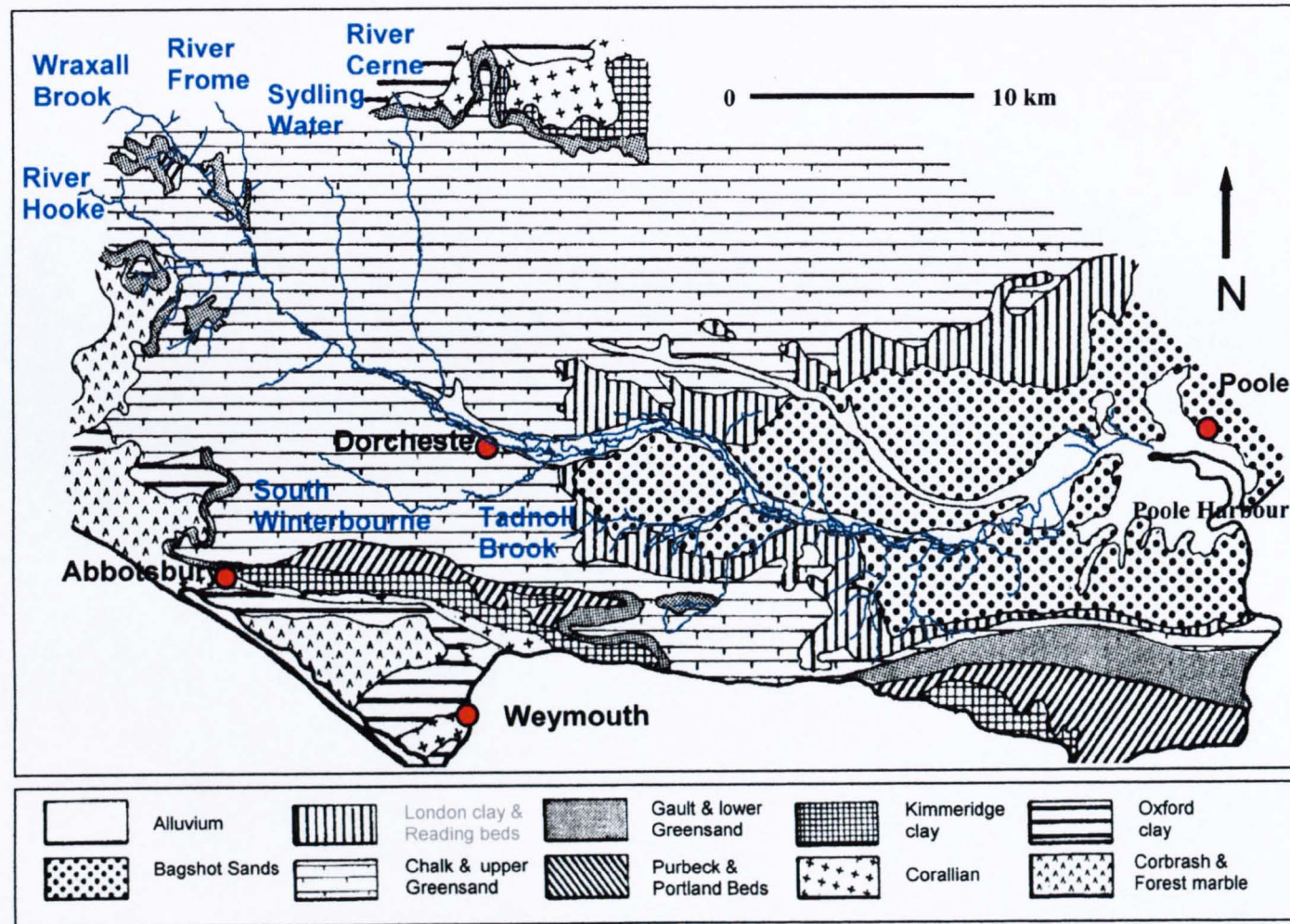


Figure 2.2 Geological map of the Frome and Piddle catchments (adapted from Paolillo (1969)). The position of the two main rivers is marked by the alluvium deposits, which extend in two lines towards Poole Harbour. Chalk outcrops over 46% of the Frome catchment.

The Frome is considered a large chalk river, but, like many chalk rivers, it has a low drainage density (Wright, 2003). There are six major tributaries, four of which (the Wraxall Brook, Sydling Water, the South Winterbourne and the River Cerne) flow over chalk rock, while the River Hooke catchment is mainly underlain by Greensands and the Tadnoll Brook flows over tertiary gravels (see Figure 2.2).

What follows below is a discussion of the general characteristics of English chalk streams, as described in the literature, and a comparison of these typical indicators to the physical characteristics of the River Frome.

2.2.1 The Chalk-river regime

The porosity of chalk rock means that chalk rivers receive little surface run off and groundwater flows almost entirely dominate river inputs (Berrie, 1992; Wheater *et al.* 2005). Chalk streams have the most stable flows of any river type in Britain (Haslam, 1982) and typically exhibit a regular and predictable annual hydrograph with relatively small differences between winter and summer flows. Discharge generally increases in December and continues to rise until March or April before declining steadily again until the next December (Berrie, 1992). Daily fluctuations in rainfall introduce some short-term variation in the chalk stream's regime, but spate conditions do not occur (Berrie, 1992) and major flooding events are rare (Bradford, 2002 and Westlake *et al.* 1972). The stormflow component of streamflow can be as little as 2 % of the rainfall input (Wheater *et al.*, 2007). Most chalk rivers are moderately fast flowing (typically $0.1 - 1.0 \text{ m s}^{-1}$; Berrie, 1992), owing to their relatively steep gradient, but they produce flat flow duration curves that reflect low

variability about the mean (Bradford, 2002). The groundwater response may be modified in chalk catchments which contain significant impermeable surface deposits, e.g. London clay, producing a hydrological regime dominated by storm flow response (Wheater *et al.* 2005)

The outcropping of clay in the Frome catchment is insignificant compared to the dominant permeable chalk and bagshot sands (Figure 2.2) and the hydrological regime for the River Frome conforms to the expected annual pattern of chalk rivers: discharge increases quickly in the autumn to reach a maximum and then slowly decreases until late summer (Dawson, 1976). Paolillo (1969) presented annual hydrographs for six consecutive years (1961 – 1966), which clearly illustrated the seasonal rise and fall of flow volume in the River Frome (see Figure 2.3).

2.2.2 Sediment in chalk streams

The bed substrate of chalk rivers generally reflects the stability of their hydrological regime. In common with many lowland rivers, chalk streams do not usually possess enough energy to move gravel and stones (Haslam, 1982) and generally have coarse gravel and cobble beds. In addition, the gravel and cobbles are often cemented together by calcareous deposits, which further reduces their transport potential. Sand and silt layers may be deposited above the gravel base but accumulation is very low and fine sediment is usually confined to discrete deposits beneath plant beds. Little silt is weathered from the chalk rock, and most silt is organic in origin (Haslam, 1982).

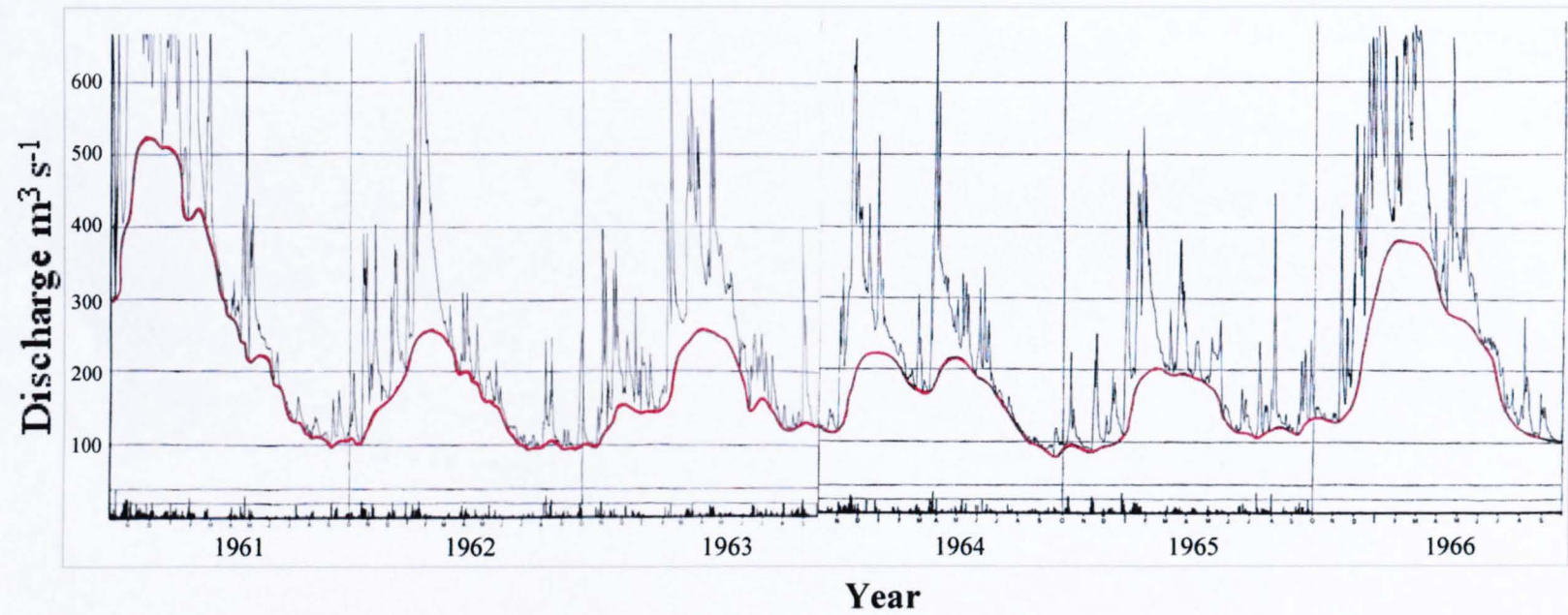


Figure 2.3 The chalk river regime: seasonal flow variations in the River Frome, 1961 – 1966. Adapted from Paolillo (1969).

Descriptions of the bed substrate of the Frome are sparse. Dawson (1976) describes the lower sections of the Frome below Dorchester, stating that the Frome here exhibits slow, silted sections behind weirs contrasting with fast gravelly riffles in between, which is in broad agreement with the expected substrate of a Chalk stream. Westlake describes the bed substrate of the Frome as consisting of “coarse gravel containing many flints, often cemented by calcareous deposits” (Westlake, 1968, p 618). Westlake also reports that sand, silt and clay are often deposited above the gravel base, especially within plant beds in summer.

2.2.3 Water quality in chalk streams

The water in chalk streams reflects its calcareous origins and has a high pH and a high ionic concentration. pH has been variously quoted as ranging from 7.4 - 8.0 (Berrie, 1992) and 7.5 - 8.5 (Westlake *et al.* 1975), while conductivity is estimated as being between 350 and 588 $\mu\text{mho/cm}$ (with a 25°C reference) (Westlake *et al.*, 1975). Chalk rivers are also renowned for their “sparkling clarity” (Haslam, 1982) and are not often turbid (Heywood and Walling, 2003), especially during the summer period of plant growth. Turbidity may increase during early winter when fine sediment is washed from beneath vegetation following senescence (Westlake *et al.*, 1972). Concentrations of the major plant nutrients (nitrate, phosphate, potassium and silicate) are all thought to be well above the critical levels required for optimal plant growth (Westlake *et al.*, 1972 and Berrie, 1992) and temperatures in chalk rivers remain surprisingly constant; leading to high levels of plant growth. Groundwater emerges from the Chalk aquifer at a stable 11°C throughout the year and this has a moderating effect on annual temperature ranges: the groundwater influence has a

warming effect during winter and a cooling effect in summer (Berrie, 1992). Thommen and Westlake (1981) estimate a maximum seasonal range of 2 – 25°C in the main reaches of chalk rivers and a monthly mean variation of less than 8°C.

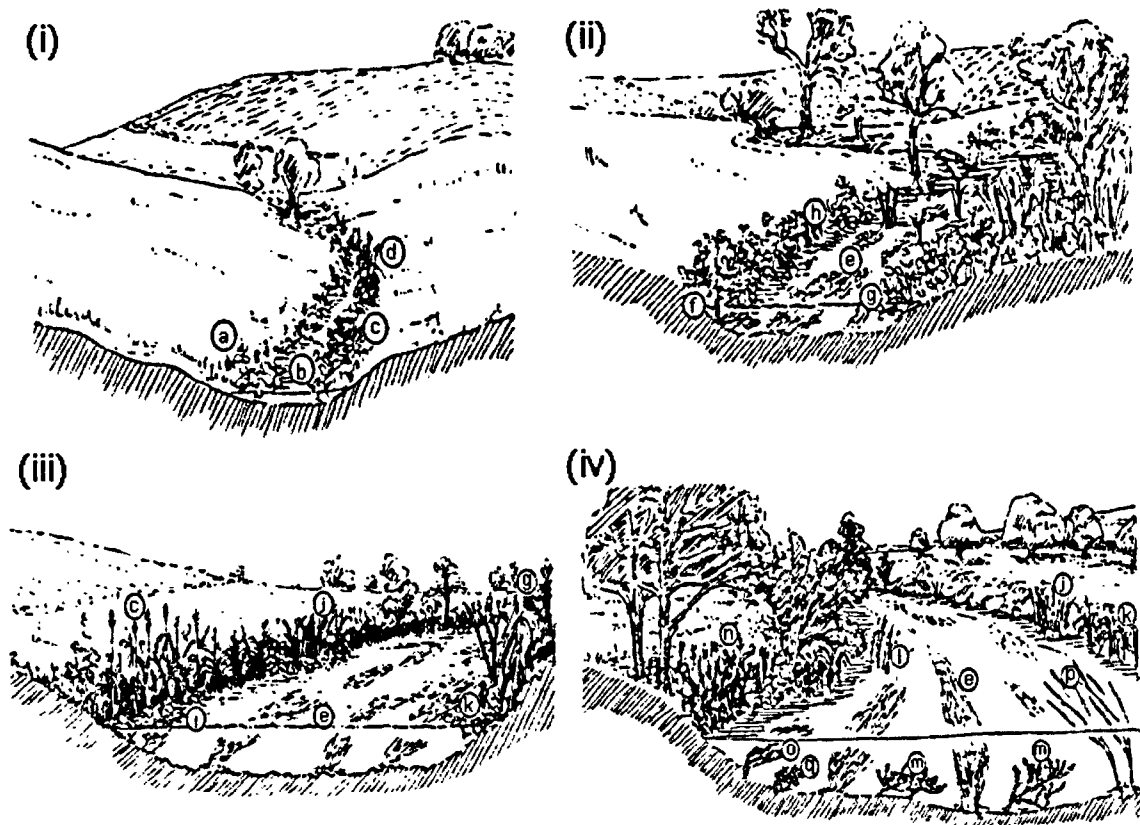
The Frome would seem to follow these moderated trends. Westlake (1972) provides a comprehensive summary of the water quality indices of the Frome as recorded at East Stoke from 1965 – 1970. The reported pH at East Stoke was 7.8 – 8.4 which tallies well with the chalk averages reported by Westlake *et al.* (1975) and Berrie (1992). Westlake (1968) reports that levels of phosphate and nitrogen in the Frome were high (30 – 160 mg P/l and 1- 3.4 mg N/l respectively) and, additionally, that dissolved oxygen levels varied between 75 - 165% saturation. Casey (1973) further affirms this, stating that an excess of plant nutrients exists in the Frome. Westlake also describes the temperature range in the Frome, reporting a range from 5.5 – 7.5 °C in January to 16.0 – 18.5 °C in August, while Dawson (1976) is in agreement with this and reports a temperature range for the Frome (1969 – 1972) as 5 – 15 °C. Both these quoted ranges lie well within the temperatures reported above by Thommen and Westlake (1981).

2.2.4 Chalk stream flora

The high levels of plant nutrients in chalk streams, combined with stable river flows and controlled temperatures, provide optimal growth conditions for many plant species. British chalk streams are known to support diverse and productive communities of high biomass (Berrie, 1992) and summer plant biomass may reach 400g dry weight m⁻² in some chalk streams. Average biomass is typically much

lower at around 200g dry weight m⁻² (Berrie, 1992). The diversity and abundance of vegetation at a specific reach depends on a) the size of the stream and b) the longitudinal position of the reach within the river. The flora of a chalk river changes with distance downstream, becoming more diverse with increasing distance from the source (Dawson, 1976). At its headwaters, a chalk stream is usually dominated by riparian and emergent species, such as *Apium nodiflorum* (fool's water cress), *Rorripa nasturtium-aquaticum* (water cress) and, occasionally, *Berula erecta* (water parsnip). Downstream, however, it is more usual for the submergent *Ranunculus* subgenus *Batrachium* (water crowfoot) to predominate, with *Ranunculus calcareous* generally the most abundant species (Dawson, 1976; Berrie, 1992). Haslam and Wolsley (1981) presented a comprehensive description of four types of chalk stream based on river dimensions and vegetation characteristics: (i) small streams without water supported species; ii) small streams with water supported species; iii) medium streams; and iv) large streams. These are illustrated in Figure 2.4. These four river types could also be considered to form a continuum from source to mouth in large chalk streams such as the River Frome.

The first of Haslam and Wolsley's four river groups: 'Small streams without water supported species' are localised in extent. They are characterised by the upper, winterbourne, portions of chalk rivers and are dry for most of the year and limited to 1-3 m in width. The probable species list for this river type is dominated by marginal and riparian plants and includes *Mentha aquatica* (Watermint), *Myosotis scorpioides* (Forget me not), *Phalaris arundinacea* (Canary Grass), *Veronica anagallis-aquatica* (Water Speedwell) and *Veronica beccabunga* (Brooklime). The second category: 'Small streams with water supported species' are much more common and occur



Key:

- | | |
|--------------------------------|------------------------------------|
| a. <i>Mentha Aquatica</i> | j. <i>Sparganium erectum</i> |
| b. <i>Mysotis scorpiodes</i> | k. <i>Schoenoplectus lacustris</i> |
| c. <i>Phalaris arundinacea</i> | l. <i>Carex acutiformis</i> |
| d. <i>Veronica beccabunga</i> | m. <i>Elodea Canadensis</i> |
| e. <i>Ranunculus</i> spp. | n. <i>Glyceria maxima</i> |
| f. <i>Berula erecta</i> | o. <i>Oenanthe fluviatilis</i> |
| g. <i>Rorripa</i> spp. | p. <i>Sparganium emersum</i> |
| h. <i>Catabrosa aquatica</i> | q. <i>Zannichelia palustris</i> |
| i. <i>Callitriche</i> spp. | |

Figure 2.4 Illustrations of the four river types from Haslam and Wosley's (1981) floristic classification of chalk river reaches: (i) small streams without water supported species; (ii) small streams with water supported species; (iii) medium streams; and (iv) large streams. Selected representative macrophytes species have also been depicted.

slightly further downstream. They may experience dry spells in late summer, but more typically they maintain a shallow (20 - 40 cm deep), perennial flow. They usually contain many of the marginal species associated with the drier winterbourne streams but they are also populated by the emergents *Apium nodiflorum*, *Berula erecta*, *Rorripa nasturtium-aquaticum* and *Sparganium erectum* (Branched Bur-reed). They can also support submerged plants such as *Callitriche* spp. (Water Starwort) and shorter-leaved *Ranunculus* spp.

The third group: 'Medium streams' usually occur much further down the catchment and, as suggested, they are wider (4 – 8 m) and deeper (30 – 75 cm) than the previous river types. They usually experience moderate flows and have a gravel bed with little silt. They are dominated mid-stream by submerged *Ranunculus* spp., particularly the longer-leaved *Ranunculus calcareous*, while *Sparganium erectum*, *Phalaris arundinacea*, and *Carex acutiformis* (Lesser Pond-sedge) populate more marginal areas. The fourth and final river group: 'large streams' are infrequent within the chalk context and are generally greater than 10 m in width and more than 75 cm deep. The flow here is slow to moderate and they may experience some silting. Again, the dominant macrophytes are the longer-leaved *Ranunculus* spp., most often *Ranunculus calcareous*. The same marginal plants are present as in medium streams but new mid-channel submergent species appear in the slower-moving water. These include *Sparganium emersum* (Unbranched Bur-reed), *Oenanthe fluviatilis* (River Water-dropwort) and the introduced *Elodea canadensis* (Canadian Pondweed).

Haslam (1982) describes the Frome's vegetation as generally similar to other chalk streams but also suggests that other rocks within the catchment may have an influence on river flora. Overall, Haslam (1982) describes the Frome as rich in species and suggests that the non-chalk influence is exemplified by a higher proportion of semi-eutrophic species, which are observed further upstream than expected. Dawson (1976) gives a comprehensive account of vegetation biomass in the Frome: he describes the average July biomass at East Stoke (1969) as $202 \pm 72 \text{ g m}^{-2}$ dry weight and also describes biomass with relation to depth: biomass in waters dominated by *Ranunculus calcareous* ranged from as little as 30 g m^{-2} in deep water ($> 1 \text{ m}$) to $300 - 400 \text{ g m}^{-2}$ in shallow water ($< 1 \text{ m}$). The average for deep and shallow waters combined was quoted as 200 g m^{-2} . Westlake (1975) also describes the biomass of the Frome and quotes an average spring biomass of $127 - 240 \text{ g m}^{-2}$ (1967 – 1968), which is similar to Dawson's 1976 estimate. These figures compare well with the average chalk stream biomass quoted by Berrie (1992). The longitudinal progression of chalk river types on the River Frome, based on Haslam and Wolsley's classification, is explored in detail in Section 2.3 of this chapter.

2.2.5 Chalk stream fauna

The variety and abundance of plants in chalk rivers provides many and diverse niche habitats for river fauna; especially since, as 'ecological engineers' (Jones *et al.*, 1994), aquatic plants are capable of altering velocity and sedimentation patterns and creating more diverse physical habitats. Each plant species supports a specific faunal assemblage. For example, Mantle and Mantle (1992) describe the dominant chalk stream macrophyte, *Ranunculus calcareous*, as sustaining large populations of

Chironomid larvae (non-biting midge), *Similium* (black fly larvae) and *Baetis* (olive mayfly) while *Nasturtium officinale* (watercress) supports *Gammarus pulex* (freshwater shrimp) and *Tubificidae* (worms). Wright (1982) states that chalk stream biota reaches high levels of abundance both in terms of numbers and biomass. Chalk rivers often support valuable fisheries and are known to be especially suitable for *Salmo salar* (Atlantic Salmon) and *Salmo trutta* (Brown trout) (Berrie, 1992 and Wright, 2003). The management of chalk streams is often geared towards these economically important fish species.

Wright (2003) describes the dominant and notable fauna of Dorset rivers, including the River Frome, and states that there are several hundred species of invertebrates in Dorset rivers covering the full range of functional feeding groups. Wright also describes the fish population of the Frome which is dominated by *Salmo salar*; *Salmo trutta*; *Dicentrarchus labrax* (Bass) and *Chelon* spp. (Mullet).

2.2.6 Management and human impact in chalk streams

Chalk streams are generally of very high quality. The UK Biodiversity Action Plan Steering Group for Chalk Rivers (2004) report that at least 50% of chalk rivers have high or very high channel habitat quality (1994 – 1997) and that, in 2000, 89% of chalk rivers were graded as good to very good in biological quality and 83% were of good or very good chemical quality. However, chalk streams have a long history of human impact and few, if any, are truly natural.

Early management of chalk streams involved the clearing of riparian tree cover and the construction of weirs, hatches, and supply channels to drive water mills and to irrigate water meadows (Westlake *et al.*, 1972; Berrie, 1992). As a result of this human management, most chalk rivers exhibit a characteristically braided morphology in their lower reaches and possess multiple diverging and converging channel threads (Haslam, 1982). Most of the supply channels and irrigation networks are now defunct, though some are still used for fisheries management. Even when derelict, however, the redundant channel networks and remnants of infrastructure will affect river flows and continue to alter the course of many rivers (Westlake *et al.*, 1972).

With the closure of water mills and the abandonment of the water-meadow systems, modern economic interests in chalk rivers have switched to fisheries and commercial watercress farming. Chalk stream fisheries are extremely valuable and fish farms are common on some rivers (Berrie, 1992) but the main emphasis is on recreational fly-fishing (Haslam, 1982). A complex system of vegetation and river management has been developed to help optimise fish populations and this involves the control of water velocity and water depth and the manipulation of vegetation and sediments. Aquatic plants in chalk rivers are regularly cut for both fisheries and flood prevention, often once or twice in every year and usually in late spring and late summer (Westlake *et al.*, 1972; Berrie, 1992) and concreted sediments are blasted with high pressure water jets to break up the calcium deposits and encourage the production of salmon redds. Another major human impact upon chalk rivers is the abstraction of groundwater for domestic water supply. Chalk aquifers provide 53%

of the UK's water needs (Bradford, 2002) and the unsustainable abstraction of groundwater can lead to reduced river discharges.

The River Frome has been subject to each of the management pressures described above. The Frome is braided for around two thirds of its length, and though some channels are natural, these networks mainly consist of flood relief channels, remnants of supply channels for mills in Dorchester and Wool and extensive water meadow networks (Environment Agency, 2005). The Frome has also been affected by watercress farms and fisheries; Wright (2003) identified nine separate cress farms within the Frome and Piddle catchments and the Environment Agency (2005) describes salmon and trout angling on the River Frome. Wright (2003) also describes an increasing demand for water resources in Dorset to be met by both groundwater and surface water abstraction. There are currently 308 abstraction licenses approved in the Frome, Piddle and Purbeck catchments (Environment Agency, 2005). Wright refers to the detrimental effects of water abstraction on both the River Allen and River Piddle, but does not identify any problems in the River Frome. Similarly, the Environment Agency has classified the Frome as 'water available' meaning that there is a "surplus of flow above that required by the environment" (Environment Agency, 2005 p.28) and potential for further abstraction.

2.3 Reconnaissance of the River Frome catchment

A rapid reconnaissance of the Frome catchment was carried out in May 2005; timed to coincide with the flowering of *Ranunculus* spp. and predicted maximum *Ranunculus* biomass in the river (Dawson, 1976; 1980). Forty sampling sites were

chosen, encompassing the entire length of the River Frome and four of its main tributaries. Sites were sampled using bridge and fording points; twenty two sites were visited on the Frome itself and eighteen on the major tributaries: the Wraxall Brook; the River Hooke; the River Cerne and Sydling Water. Several photographs were taken at each site to visually record changes in channel dimensions, channel planform, aquatic vegetation type, riparian land use, and any evidence of management or artificial controls. In addition, the width and depth of the channel at each site was quantitatively estimated in the field, and notes were taken on any points of interest such as the presence of weir hatches; braided channels and scientific monitoring equipment. The location of each sampling site is illustrated in the pullout map in Figure 2.5 and photographs of selected reaches are displayed in Plate 2.1. The Dorchester and Weymouth Landranger OS map was used to select accessible sites. These were mainly located at bridge and fording points, where the river could either be viewed from above or easily navigated on foot. The use of public sites removed the need to acquire access from multiple landowners but may have introduced an element of bias into the survey, with natural changes perhaps being obscured by human influence. Accordingly, signs of human influence were recorded in the study. The data and photographs obtained in the reconnaissance were used to examine the longitudinal changes in the river Frome from source to mouth, to place both the field sites and the River Frome in context.

2.3.1 Channel dimensions

The Photomontage in Plate 2.1 shows the change in channel dimensions from source to mouth at selected sampling sites on the River Frome. It is evident from the photos

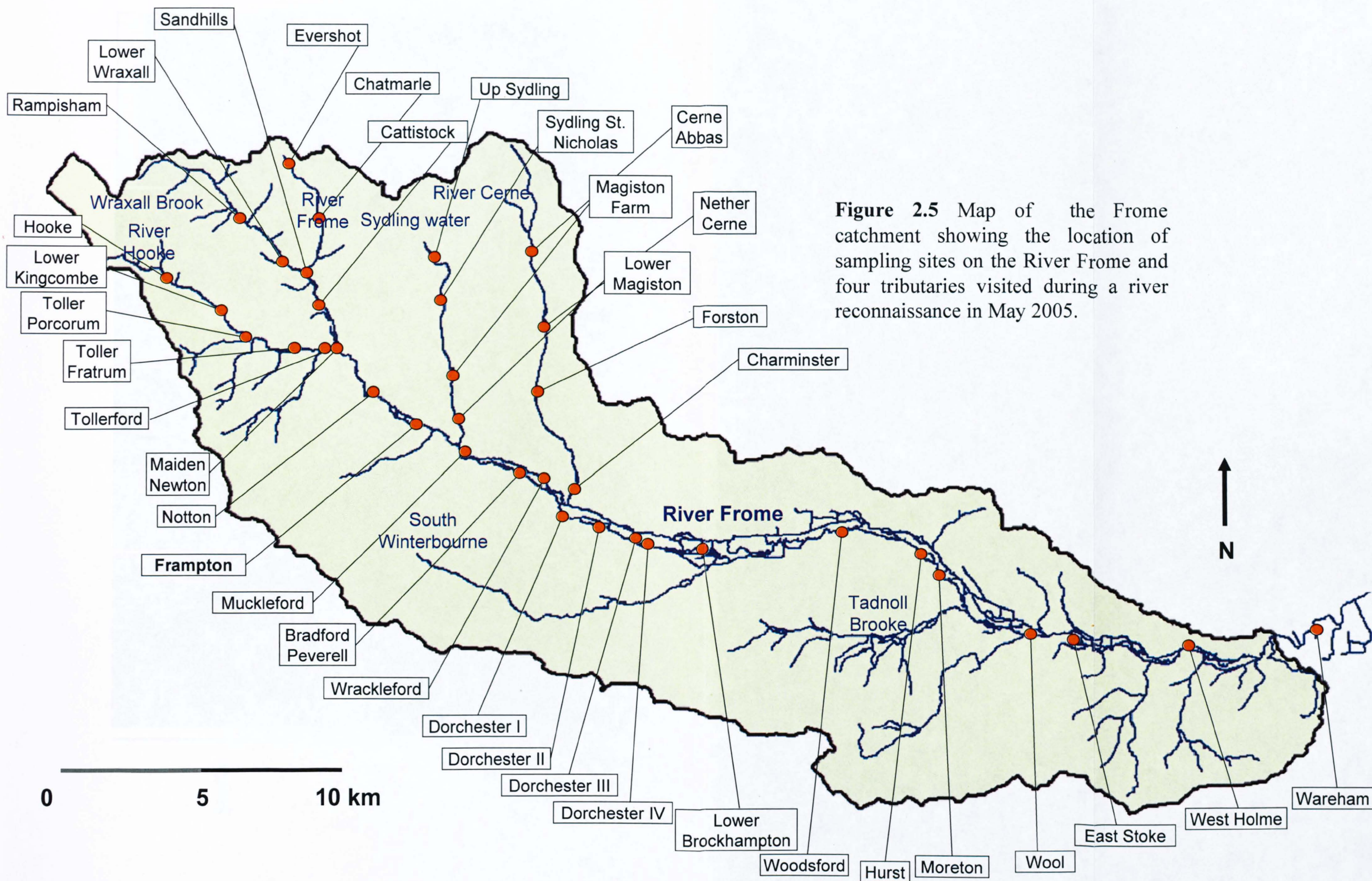
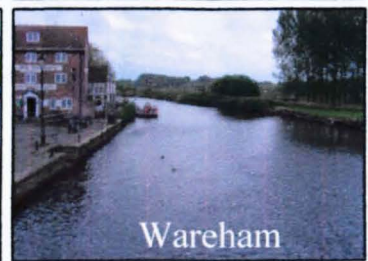
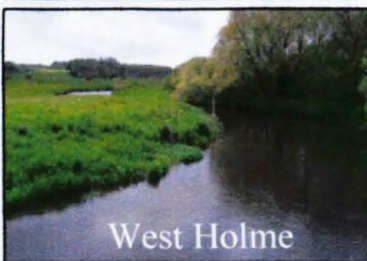
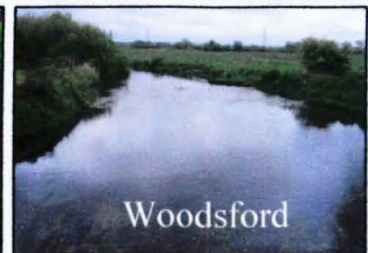
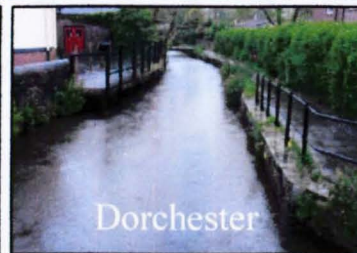
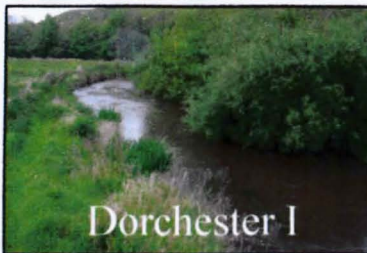
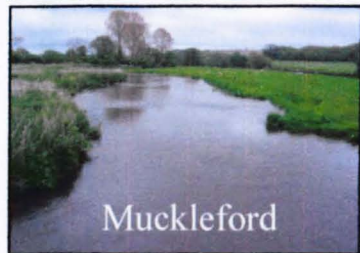
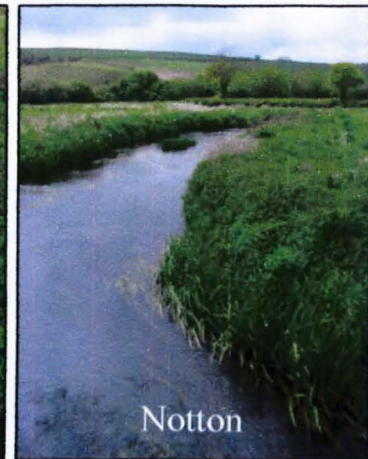
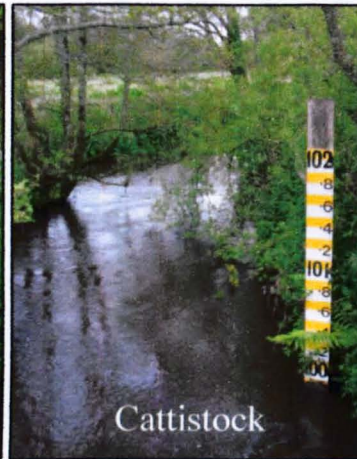
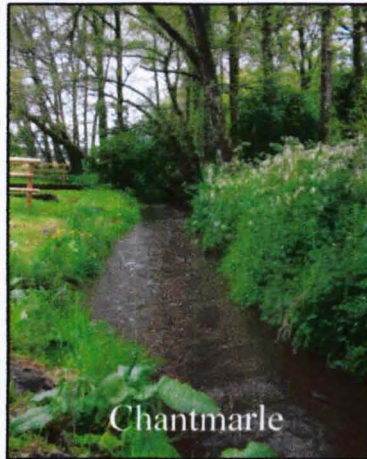
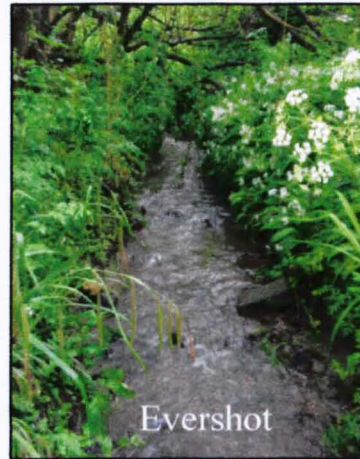


Figure 2.5 Map of the Frome catchment showing the location of sampling sites on the River Frome and four tributaries visited during a river reconnaissance in May 2005.

Source



Mouth

Plate 2.1 Photomontage of selected river reconnaissance sampling sites on the River Frome. Photographs are arranged in order from source to mouth and show clear changes in river width and depth with progressive distance downstream.

that width and depth generally increase with progressive distance downstream, but that some interruptions to this general trend also occur, e.g. at Moreton where the river width is 58 m. This photographic record was supplemented by quantitative estimates of width and depth as recorded in Table 2.2.

Width and depth measurements were used to derive the width/depth ratio, to allow comparison of the shape of channels of differing size. Figure 2.6 shows a histogram representation of the of the width/depth ratio distribution for: a) the Frome catchment; and b) the main River Frome. The plots show a restricted range of values and a slight negative skew in the distribution. This is also illustrated in Figure 2.7 which shows the changes in the width/depth ratio from source to mouth on the main river Frome and the sampled tributaries. The width/depth ratio of the Frome is relatively constant from source to mouth. Obvious exceptions exist, e.g. at Maiden Newton, Wool, and again at Moreton (a very wide and shallow fording point), but the general trend describes a relatively constant channel shape with low - moderate width/depth ratios. This is exemplified by the median width depth ratio, 5.63, which indicates that channel width on the Frome is, on average, 5-6 times the channel depth.

2.3.2 Land use and management

The Frome catchment is principally agricultural and is dominated by grassland and cereal cropland (Casey and Newton, 1973), and the land use of the sampled sites reflects the general land use for the Frome catchment as a whole. The majority of sites were in agricultural grassland (65%), both improved grassland (35%) and rough

Table 2.2 A classification of sites sampled during a catchment reconnaissance of the River Frome and four of its major tributaries. Classifications are based on Haslam and Wolsley's (1981) typology of chalk streams.width

River Type	Sites location on Frome and 4 major tributaries	Notable Predicted Species (Haslam and Wolsley (1981))	Notable Observed Species	Channel Width & Depth (m)
1. 'Small streams without water supported species'	Evershot Chantmarle	<i>Mentha aquatica</i> <i>Myosotis scorpioides</i> <i>Phalaris arundinacea</i> <i>Veronica anagallis-aquatica</i> <i>Veronica beccabunga</i>	<i>Grasses</i> <i>Berula erecta</i> <i>Athyrium spp.</i>	1 × 0.5 1.5 × 0.5
2. 'Small streams with water supported species'	Cattistock Rampisham Lower Wraxall Sandhills Hooke Lower Kingcombe Toller Porcorum Toller Fratrum Tollerford Cerne Abbas Nether Cerne Forston Charminster Up Sydling (i) Up Sydling (ii) Sydling St Nicholas Magiston Farm Lower magiston	<i>Apium nodiflorum</i> <i>Berula erecta</i> <i>Rorripa nasturtium-aquaticum</i> <i>Sparganium erectum</i> <i>Callitriche spp.</i> <i>Ranunculus spp</i>	<i>Berula erecta</i> <i>Rumex spp.</i> <i>Veronica beccabunga</i> <i>Sparganium erectum</i> <i>Petasites hybridus</i> <i>Symphytum asperum</i> <i>Ranunculus spp.</i> <i>Rorripa nasturtium-aquaticum</i> <i>Fontinalis</i> <i>antiptretica</i> <i>Iris pseudacorus</i> <i>Urtica dioica</i>	6 × 1.5 3.5 × 1.25 5 × 1.1 4.5 × 0.6 2.25 × 0.45 3 × 0.65 3 × 1 7.5 × 1 4 × 0.8 4 × 0.7 2.5 × 0.45 4 × 1.5 4 × 0.6 1 × 0.3 3.5 × 0.4 2.5 × 1 2 × 1 3 × 0.6
3. 'Medium streams'	Maiden Newton Notton Frampton Muckleford Bradford Peverell Wrackleford Dorchester (i) Dorchester (ii) Dorchester (iii) Dorchester (iv) Lower Brockhampton	<i>Ranunculus calcareous</i> <i>Sparganium erectum</i> <i>Phalaris arundinacea</i> <i>Carex acutiformis</i>	<i>Ranunculus calcareous</i> <i>Berula erecta</i> <i>Urtica dioica</i> <i>Phalaris arundinacea</i> <i>Rorripa nasturtium-aquaticum</i> <i>Sparganium erectum</i> <i>Symphytum asperum</i> <i>Veronica</i> <i>Beccabunga</i>	8.5 × 0.4 7.5 × 1.1 14 × 0.8 14 × 1.35 9.5 × 1.5 8 × 1 * * 11 × 1.5 18 × 1 14 × 1
4. 'Large streams'	Woodsford Hurst Moreton Wool East Stoke West Holme Wareham (i) Wareham (ii)	<i>Sparganium emersum</i> <i>Oenanthe fluviatilis</i> <i>Elodea canadensis</i>	<i>Ranunculus</i> <i>Calcareous</i> <i>Urtica dioica</i> <i>Rorripa nasturtium-aquaticum</i> <i>Phalaris arundinacea</i>	16.5 × 2 22.5 × 1.5 58 × 0.35 24.5 × 0.4 12 × 1 32 × 3 21.5 × 6 44 × 4.5

* Sampling of width and depth not possible due to site conditions.

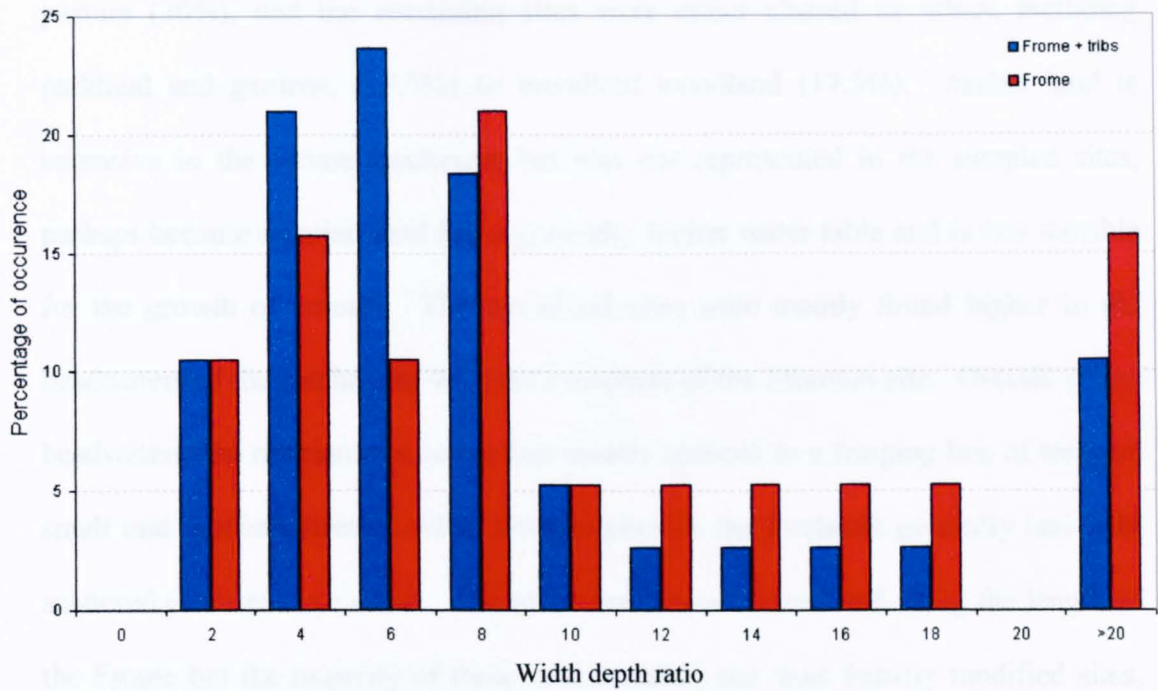


Figure 2.6 Multiple histogram comparing the distribution in width/depth ratio for sampled sites on the River Frome and sites sampled from the Frome and four of its major tributaries.

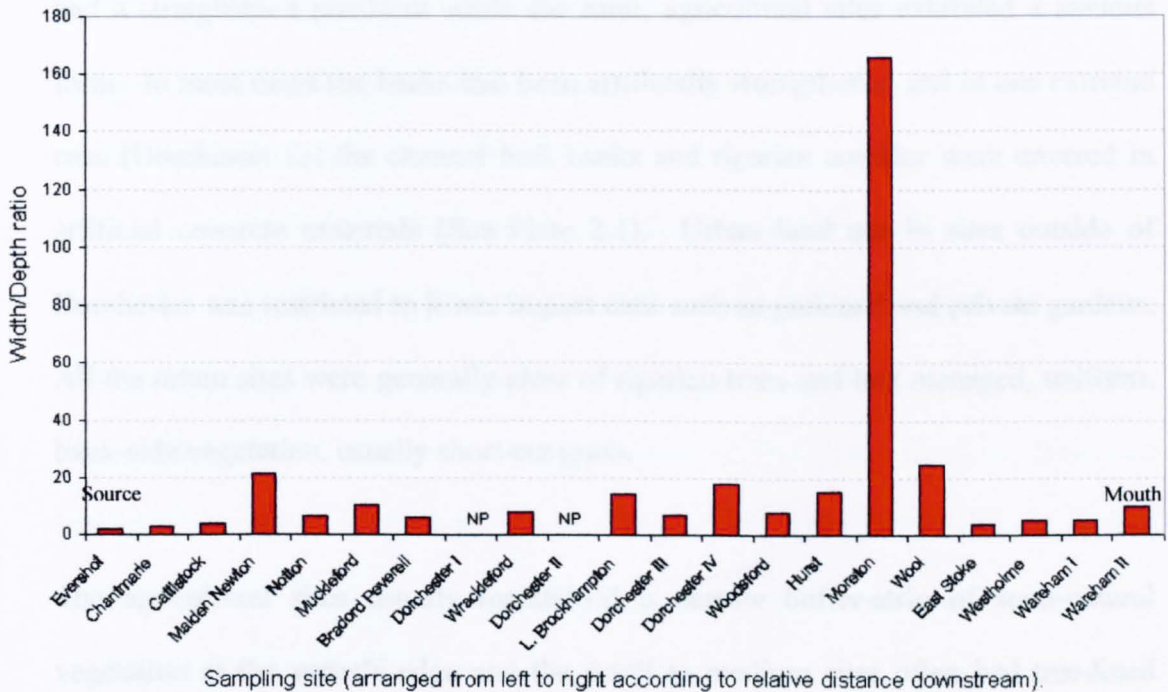


Figure 2.7 Bar chart showing the width/depth ratio of sampled sites on the River Frome arranged from left to right according to relative distance downstream.

pasture (30%), and the remaining sites were either classed as urban, including parkland and gardens, (17.5%) or broadleaf woodland (17.5%). Arable land is extensive in the Frome catchment but was not represented in the sampled sites, perhaps because riparian land has a generally higher water table and is less suitable for the growth of cereals. The woodland sites were mainly found higher in the headwaters of the catchment, with the exception of the Moreton site. Outside of the headwaters, the riparian tree cover was usually reduced to a fringing line of trees on small and medium streams while river reaches in the lowlands generally had only scattered or absent tree cover. The urban reaches were scattered along the length of the Frome but the majority of these, and certainly the most heavily modified sites, were located in Dorchester. Artificial modifications within the urban areas were most evident from changes in the channel planform. All the sites classed as urban had a straightened planform while the rural, agricultural sites exhibited a sinuous form. In most cases the banks had been artificially strengthened and in one extreme case (Dorchester iii) the channel bed, banks and riparian corridor were covered in artificial concrete materials (See Plate 2.1). Urban land use in sites outside of Dorchester was restricted to lower impact uses such as parkland and private gardens. All the urban sites were generally clear of riparian trees and had managed, uniform, bank-side vegetation, usually short-cut grass.

The agricultural sites usually maintained a narrow buffer-strip of semi-natural vegetation at the water's edge and the small to medium sites often had tree-lined banks. The water-meadow system was evident on many of the lower sites on the Frome. For example, the sites at Lower Brockhampton and Dorchester (iii) possess two or more separate channel threads and a disused weir hatch was observed at

Forston on the River Cerne. Scientific monitoring stations were noted at Cattistock and at East Stoke.

2.3.3 Macrophytic and riparian vegetation

In terms of its vegetation, the River Frome appears to consist of a continuum of river types from source to mouth. A four-part index of chalk rivers, developed by Haslam and Wolsley (1981), was utilised to classify each sample reach and to compare observed and predicted plant species. The site classifications for the River Frome reconnaissance sites are summarised in Table 2.2.

The sampled sites for the Frome at Evershot and Chantmarle appear to conform floristically to Haslam's type 1 chalk stream: 'Small streams without water supported species'. No 'water supported' submergent or emergent plants are present at these two sites and the riparian plants are dominated by riparian grasses and *Berula erecta* with some *Athyrium* spp. (Fern) and *Rumex* spp. (Dock). The headwater sites of the four main tributaries, Rampisham, Hooke, Cerne Abbas and Up Sydling, and the third site on the Frome, Cattistock, seem to conform more to Haslam's second river type: 'Small streams with water supported species'. Here the same riparian species dominate the banks, especially *Berula erecta*, but emergent species are also represented. The dominant emergent macrophytes are *Veronica anagallis-aquatica*, *Veronica beccabunga*, *Rorippa nasturtium-aquaticum* and *Sparganium erectum*. Sydling Water was unusual within the headwater reaches as it contains dense stands of submerged *Ranunculus* spp. but this was a densely packed and short-stemmed variety (Plate 2.2).

The lower sites on the Frome from Maiden Newton to Lower Brockhampton conform to the third chalk-stream type described by Haslam and Wolsley: 'medium chalk streams'. They are characterised by a width of less than 10 m and large swathes of long-leaved *Ranunculus calcareous* (see Plate 2.2 a) and *Phalaris arundinacea*. The 'large stream' sites, Haslam and Wolsley's (1981) fourth river type, occur on the Frome between Woodsford and Wareham at the river's mouth, and, like medium rivers, they are dominated by *Ranunculus* spp. but are generally greater than 10 m in width and 0.75m in depth. Unfortunately, due to the greater width and depth, the reconnaissance photographs do not provide much information about the submerged species at the large river sites. The presence of *Ranunculus* spp. can still be discerned at the water surface, but submerged plants that do not reach the water surface could not be identified. Haslam and Wolsley (1981) predicted an increase in submerged-species diversity in large chalk streams and this is likely to be the case for the River Frome, but this could not be confirmed from the present analysis.

In general, the species diversity and abundance of the Frome increased with distance downstream, in agreement with Dawson (1976) and Berrie (1992) in their descriptions of chalk rivers. The obvious exception to this was in urban areas. Here, species diversity was significantly reduced, either through vegetation management or because of the use of artificial bank and bed materials, e.g. at Dorchester (iii). All four types of chalk river identified by Haslam and Wolsley were present on the Frome and in the expected proportions. Type 1 reaches were present but infrequent; Type 2 and Type 3 reaches were common; and large streams, Type 4, were perhaps

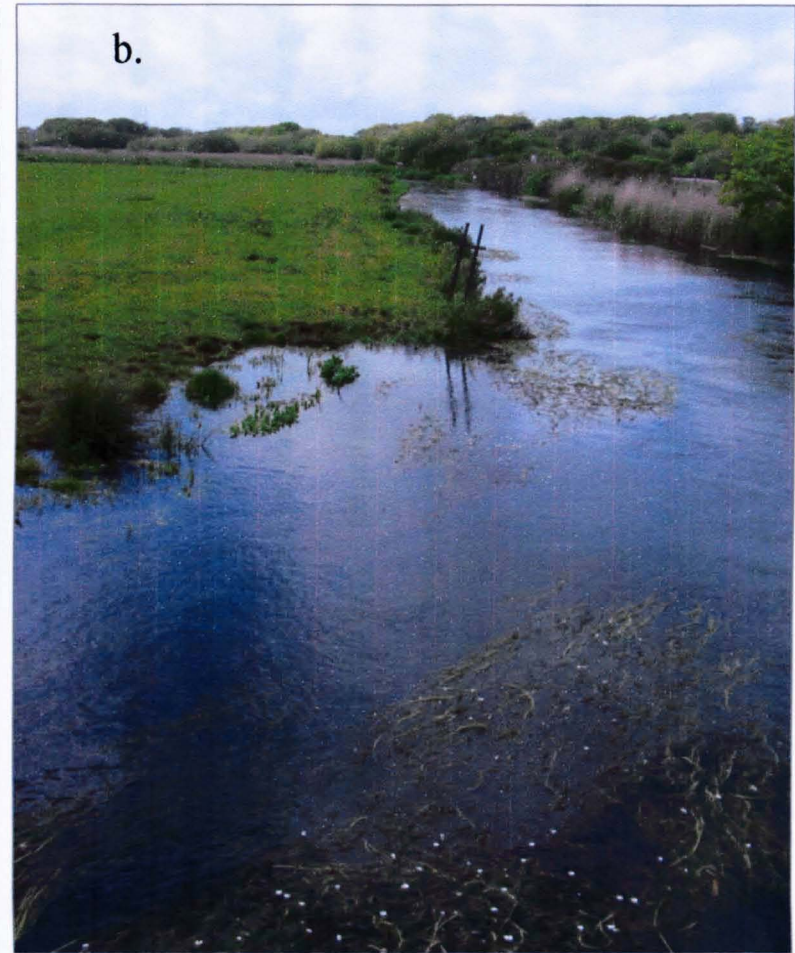
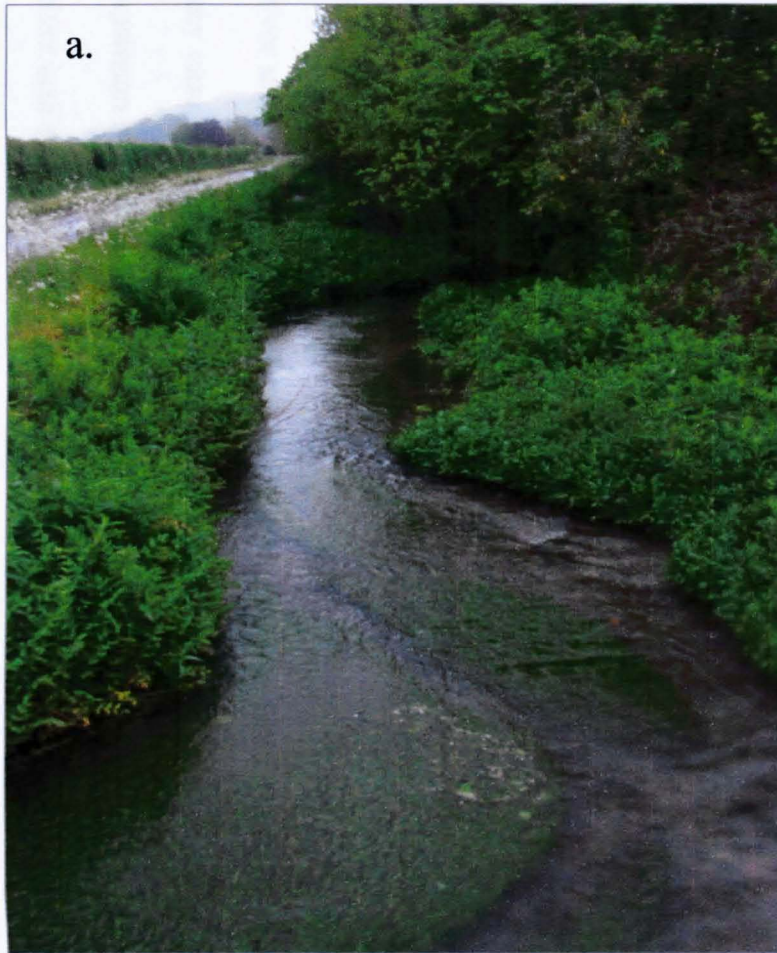


Plate 2.2 *Ranunculus* spp. in the Frome catchment (a) short-leaved species in a headwater reach at on Up Sydling on Sydling water and (b) long-leaved species on the main River Frome at East Stoke.

more common on the Frome than expected for a chalk river but were still less in number than Types 2 and 3.

2.4 Delineation of a viable stretch of the River Frome and selection of study sites

Having explored the general characteristics of the River Frome, several sites were then required for detailed investigation. The initial selection process centred on defining a viable portion of the river, as not all sections of the river would have been suitable for this study. Several factors influenced the delineation of a suitable river section and a viable stretch of river was chosen by a four-part, step-wise, elimination of less suitable reaches (see Figure 2.8).

The first major delineator was catchment geology. It was thought desirable to have the study sites located within the chalk area of the catchment, to allow more valid comparisons with other chalk streams, and to remove the influence of other rock types which may locally determine site flora (Haslam, 1982). The boundary between chalk and gravel was identified as lying just downstream of the confluence with the South Winterbourne, just West of Dorchester (see Figure 2.2) and this meant that river sections downstream of this point were not considered suitable for the study. In any case, as a second delineator, it was thought desirable to conduct any research upstream of Dorchester, to negate the influence of any pollutants or outfalls into the river and to avoid very public or heavily modified sites. Therefore, the boundary was shifted upstream so that any sites downstream of Dorchester were not considered.

The third specific delineator was channel morphology: chalk streams are noted for having a braided morphology in their lower reaches (Haslam, 1982), and there was a concern that the present study sites should be located where the Frome flows as a single channel. This would allow discharge to be calculated without the complication of several separate channel threads. After consultation of the relevant OS map, it was noted that the Frome splits into several natural and artificial braided channels after passing by the village of Stratton. Therefore, any sites below Stratton were discounted.

More generically, and as the final delineator, it was decided that the study sites should be of a width and depth which would facilitate safe working but also allow a reasonable channel area in which to take measurements. Most of the wider and deeper sections of the Frome had already been discounted through other selection criteria, but a decision was made to further restrict the study sites to sections upstream of the village of Grimstone, which is located at the confluence with Sydling water. After this, a minimum width criterion was applied and it was decided that the study sites should be at least ten metres in width. This meant that river reaches upstream of the confluences with the Wraxall Brook and River Hooke were discounted. This left a viable 8 km stretch of the Frome between the villages of Maiden Newton and Grimstone (Figure 2.8). Within this 8 km viable stretch, three sites were chosen for detailed investigation: Crockways; Hydeclift Plantation and Frampton Estate. Each of these was situated within a smaller, 2.5 kilometre section of river (GR 611 958 to GR 631 947), located close to the village of Frampton and approximately 9 kilometres northwest from the town of Dorchester. The locations of the three study sites are illustrated in Figure 2.9.

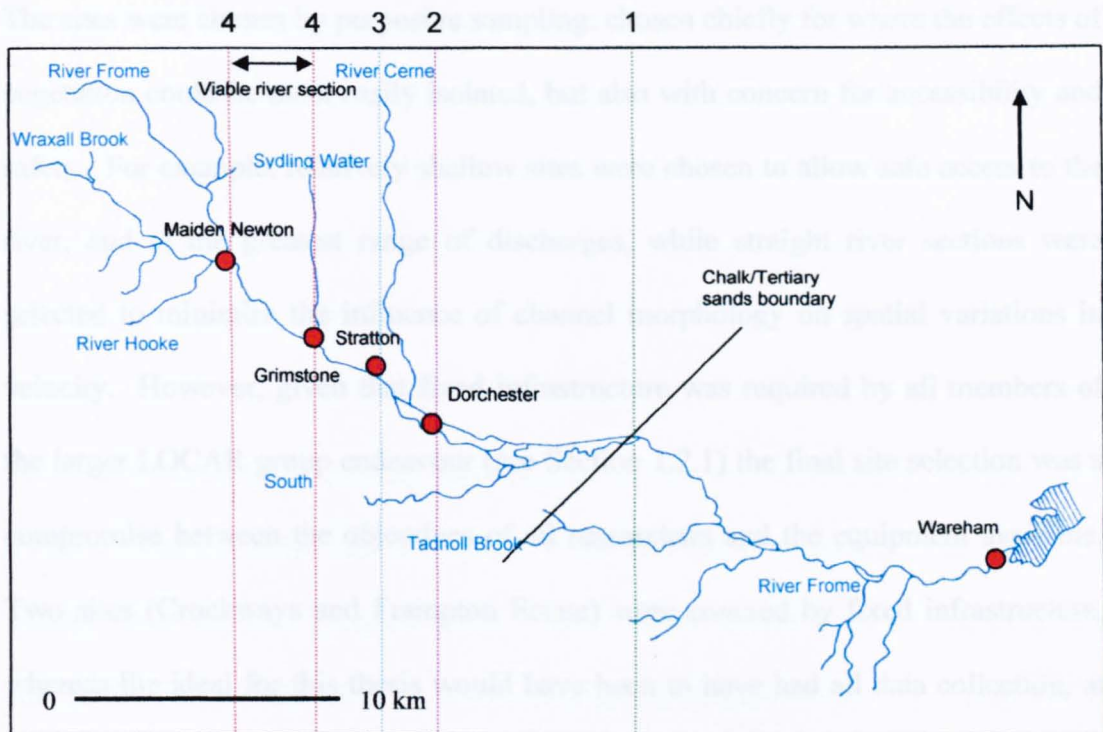


Figure 2.8 Simplified map illustrating the boundaries created by the four main selection criteria used in the choice of viable study sites on the River Frome.



Figure 2.9 Ordnance Survey map showing the viable study section of the River Frome between Maiden Newton and Grimstone. Three study sites were chosen within this viable section and the location of each site is indicated on the map.

The sites were chosen by purposive sampling: chosen chiefly for where the effects of vegetation could be most easily isolated, but also with concern for accessibility and safety. For example, relatively shallow sites were chosen to allow safe access to the river, and at the greatest range of discharges, while straight river sections were selected to minimise the influence of channel morphology on spatial variations in velocity. However, given that fixed infrastructure was required by all members of the larger LOCAR group endeavour (see Section 1.2.1) the final site selection was a compromise between the objectives of all researchers and the equipment available. Two sites (Crockways and Frampton Estate) were covered by fixed infrastructure, whereas the ideal for this thesis would have been to have had all data collection, at each of the three research scales, conducted at three contrasting sites. The three scales of research were accommodated at the same site at Crockways but not at Frampton or at Hydeclift and a compromise was made to undertake part of the data collection at Frampton and part at Hydeclift and to contrast each of these to Crockways at different research scales.

The use of three rather than two sites was necessary to provide the contrasts in land use, channel morphology and vegetation cover demanded by the thesis aims: two sites were relatively open with little tree cover (Crockways and Frampton), while at the other site the river was predominantly shaded by riparian trees (Hydeclift), and, similarly, two sites were of straight planform (Frampton and Hydeclift) while the other contained sinuous meanders (Crockways). The utilisation of each individual study site varied according to the specific features of the reach and a detailed site description and a summary of the usage of each site follows below. Each study location is first described in general terms, covering the 500m stretch defined during

primary River Habitat Surveys (RHS) (see Table 2.1), and then followed by a description of the specific PT-reaches, grid-reaches and discharge cross-sections. Finally, the infrastructure and the type of data collected at each site are listed.

2.5 Site 1: Crockways

Site 1 at Crockways is the most upstream of the three detailed study reaches. The overall character of the site is that of a series of tight meanders with relatively straight river sections between bends. River width ranges from 8 - 10 m in the straight sections to almost 15 m at the apex of the meander bends. The land use at the site is quite homogeneous and both banks are dominated by grassland, though the left bank does enter into a small area of semi-natural woodland at the end of the 500 m RHS section. The right bank is solely improved grassland and is cut for hay right to the bank edge, while the left bank is predominantly rough pasture and is grazed by livestock. In addition, the left bank possesses a fenced off, buffer strip, of semi-natural vegetation which includes tall rank herbs and scattered riparian trees. The channel morphology at Crockways is the most complex of the three study sites: two riffles and seven pools were identified within the 500m RHS reach and several islands were noted along with a point bar and several side bars. This could be ascribed to the more natural planform and relatively undisturbed channel present at Crockways, compared to the two other artificially straightened study reaches. A schematic diagram of the Crockways site is provided in Figure 2.10 and selected photos of the site are provided in Plate 2.3.

2.5.1 Description of the Crockways grid-reach

Several measurements in this study were made using a repeated sampling grid in a defined 20 m long reach (see Table 2.1) and these grid-sections require specific description. The grid-based measurements at Crockways are focused on a straight river section located in the middle of the larger 500 m reach and just upstream of the first pressure transducer (see Figure 2.10). The river here is approximately 8 – 9 metres in width with relatively high vertical earth banks, which are slightly undercut in places. The banks are largely clear of trees with only one large alder present on the right bank at the upstream margin of the grid. In summer, overhanging riparian vegetation is able to reach the water and impact on flow from the left bank only. The bed morphology at Crockways is fairly consistent throughout the grid reach, although water depth does increase slightly from the downstream to upstream end of the grid. In addition, one small pool exists at the very upstream edge of the grid; formed around the roots of the alder tree. Under the RHS classification system the grid-reach is predominated by cobble substrate (-8 to -6 phi) but in general, finer sediment is present at the channel margins, or under vegetation, and the coarser gravel material is found in the channel centre.

Notable macrophytes within the grid reach include the submerged plants *Ranunculus calcareous* (water crowfoot), *Myriophyllum alterniflorum* (water milfoil), the submerged moss *Fontinalis antipyretica* (willow moss) and the emergent macrophyte *Sparganium erectum* (branched bur-reed). *Sparganium* is confined to the channel margins in deeper, low velocity, areas while the submerged species are mainly found in mid-channel in areas of higher velocity.

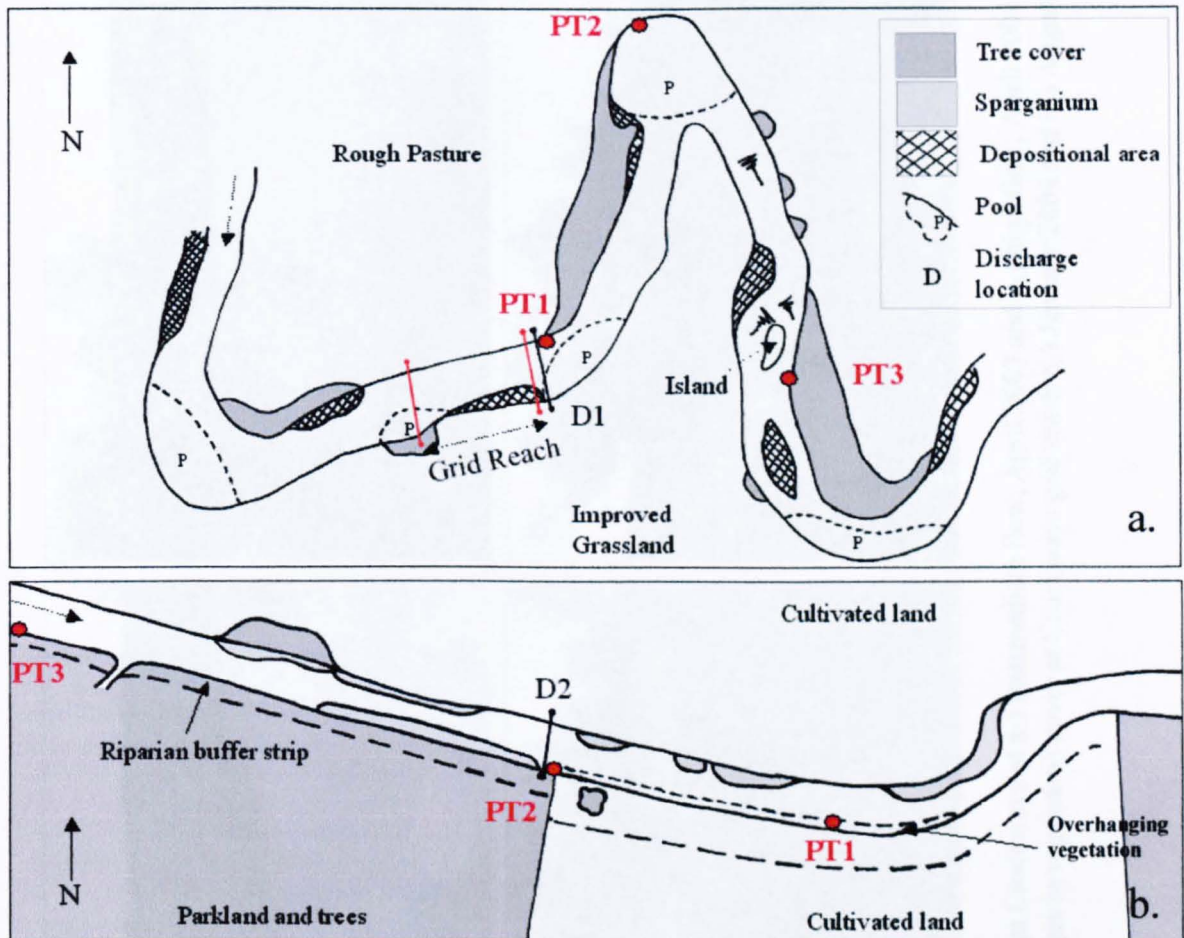


Figure 2.10 Schematic diagrams detailing the permanent monitoring infrastructure at two of the study sites on the River Frome: at (a) Crockways and (b) Frampton Estate. Diagrams are only approximately to scale. No fixed infrastructure was in place at Hydeclift given the public nature of the site.



Plate 2.3 a_1 and a_2 Meander bend at Crockways at a_1) intermediate flow, April 2003 and a_2) high flow, April 2004
Plate 2.3 b_1 and b_2 Seasonal changes in vegetation cover at Crockways grid reach b_1) January 2004 and b_2) August 2004.

2.5.2 Data collection and infrastructure at the Crockways field site

There are three pressure transducers installed on the Crockways site, located roughly equidistant along a major meander bend (Figure 2.10). The pressure transducers (PTs) give 15-minute readings of river stage which may be used to construct a water surface slope for the reach. The most upstream pressure transducer was also used as a reference against discharge measurements to construct a stage/discharge-rating curve. Discharge measurements were taken at a relatively shallow and vegetation-free cross section just upstream of PT1 and at the downstream end of the mesoscale sampling grid. The microscale measurements were undertaken within the grid reach at selected vegetation stands. The data collection undertaken at Crockways is described in Table 2.1.

2.6 Site 2: Hydeclift Plantation

Hydeclift Plantation is located roughly at the midpoint of the 2.5 km stretch between Crockways and Frampton: 1.4 km downstream from Crockways and 1.1 km upstream from the Frampton Estate. The Hydeclift study site is situated on a long, straight river section that has been artificially straightened, and possibly widened, in the past and is 10 – 16 metres wide. Land use is quite varied within the 500m stretch defined by the RHS; the right bank is improved grassland with a three-metre wide buffer strip of semi-natural vegetation, and the left bank is a mix of broadleaf woodland and parkland. The channel, however, is extremely homogeneous and there are only two identifiable riffles and one pool present within the 500m covered by the

RHS. No point or side bars were identified. Photographs of the Hydeclift study site are presented in Plate 2.4.

2.6.1 Description of the Hydeclift grid-reach

The grid-reach at Hydeclift Plantation again lay at the middle of a 500 m RHS section. It is contained within the broadleaf woodland area of the left bank, and provides a contrast with the open grid-section at Crockways. The river here is 10 – 11 metres wide and the banks are vertical or slightly undercut on the left bank and very undercut on the right bank. The left bank material is earthen while the right bank is mostly flint cobbles overlain by a thin soil layer. The left bank land use is broadleaf woodland and trees are present right at the bank edge where they are greatly undercut and roots, branches, leaves and tree trunks may at times interfere with river flow. Overhanging *Hedera helix* (English Ivy) also reaches into the channel for all of the year. The right bank is significantly higher than the left bank and only slightly undercut. The land use is improved grassland with some fringing riparian plants present at the bank edge, but little of the riparian vegetation is large enough to reach sufficiently far into the channel to have any impact on flow. Adjacent to the right bank, for part of the grid, is a side bar of fine sand and silt, which grows to support emergent vegetation in summer. The trees on the left bank extend their shade over more than half of the river channel and there is also one small hawthorn tree present on the right bank which casts shade on the top right of the grid reach. In-stream vegetation is thus more prevalent to the right of the river channel.

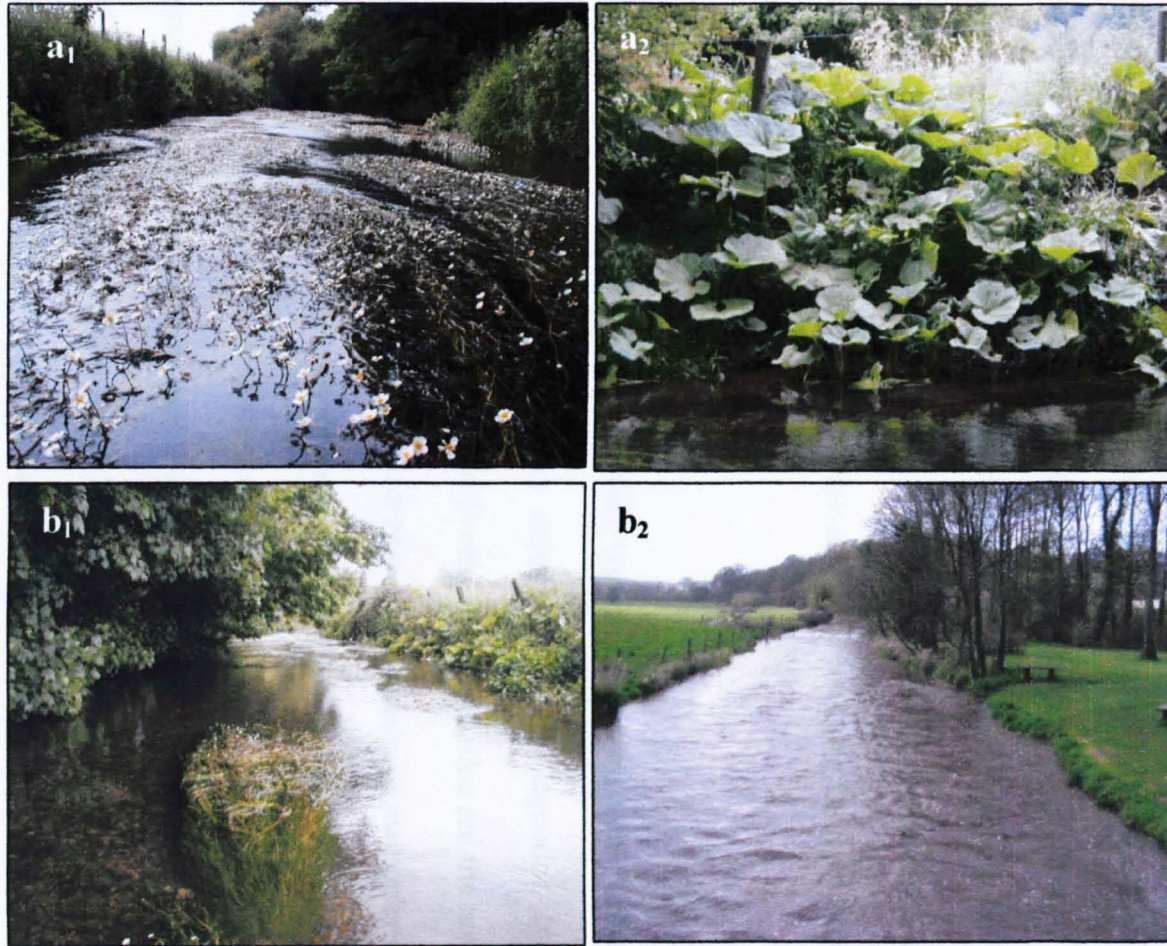


Plate 2.4 **a₁** and **a₂** Submerged and emergent macrophytes at Hydeclift Plantation: **a₁**) *Ranunculus calcareous*; and **a₂**) *Petasites hybridus*. **Plate 2.4** **b₁** and **b₂** Planform and land use at Hydeclift plantation: **b₁**) view of the shaded grid reach looking downstream; and **b₂**) improved grassland and parkland land use downstream of the grid-reach.

The bed of the grid reach is very even with only a very slight increase in water depth from the downstream to upstream end of the grid. The average velocity at Hydeclift is higher than at Crockways (24% higher at similar discharges) and, accordingly, the substrate is much coarser over most of the bed, though still classed as cobble dominated in the RHS (-6 to -8 phi). In addition, several large stones are clustered at the upstream end of the grid and some isolated large stones occur throughout. Finer sediment is present in a narrow band along the margins of the stream, particularly concentrated in the low velocity areas beneath the heavily undercut left bank, from which the sediment may well derive. Fine sediment also accumulates within plant beds during summer, especially beneath emergent herbs on the right hand margin of the stream.

Important macrophytes at the site include the large emergent herb *Petasites hybridus* (butterbur) and the smaller carpet emergent *Mentha aquatica* (water mint). *Ranunculus calcareous* and the moss *Fontinalis antipyretica* are the most abundant submerged species. *Ranunculus calcareous* is the dominant macrophyte and forms large monospecific patches several metres in length while *Fontinalis antipyretica* is mostly found attached to large stones. *Sparganium erectum* is absent from this site, possibly due to the higher velocities and coarser substrate, or perhaps due to shading of the channel. *Sparganium* does, however, appear as rare, isolated plants in its submergent form, *Sparganium emersum* (unbranched bur-reed).

2.6.2 Data collection and infrastructure at the Hydeclift Plantation site

A popular public pathway runs along the river bank at the Hydeclift site and the reach was considered too public to leave expensive equipment unattended. Therefore, no permanent infrastructure was put in place and no stage/discharge ratings were attempted. The data collection undertaken at Crockways is detailed in Table 2.1.

2.7 Site 3: Frampton Estate

The Frampton Estate site is the most downstream of the three sites, 2.5 km downstream of Crockways and 1.1 km downstream of Hydeclift Plantation. Several minor tributaries and springs feed into the river between Hydeclift and the Frampton Estate site and, accordingly, there is a higher discharge here than at the two upper sites and a greater channel width and generally greater depth. The planform is again that of a straightened channel (figure 2.10b), and the channel appears to have been over-widened in the past, at least in the upper part of the 500m reach. A natural berm runs along the left bank for approximately a fifth of the reach and indicates that the channel has readjusted to the artificial widening. In the River Habitat surveys no point or side bars were identified, though an island and three riffles and two pools were observed.

The river channel is 11 – 15 metres wide and both banks consist of vertical or undercut, earthen material. The land use on the left bank, within the 500m reach defined during the RHS, consists solely of cultivated cropland that is left bare in

winter; this land use reaches to the bank edge and there is little overhanging riparian vegetation. The right bank land use, however, is extremely varied and includes parkland, broadleaf mixed plantation, improved grassland and tilled cropland. A buffer strip of semi-natural vegetation extends along the majority of the right bank and there is intermittent tree cover on both banks.

The bed sediments at Frampton are similar to those found at Crockways and are much more diverse than at Hydeclift. The substrate ranges through a continuum from large cobbles to very fine organic sediment, but cobbles (-6 to -8 phi) were the dominant category recorded in the River Habitat Survey. Fine sediment is more abundant here than at the two upstream sites but is again concentrated at the channel margins or beneath plant beds.

The two dominant macrophytes at Frampton are *Ranunculus calcareous* and *Sparganium erectum*, both of which display luxuriant growth in summer. *Sparganium* here forms wide, near-continuous stands along both channel margins, with a few isolated stands in the middle of the channel. At the same time, interconnecting stands of *Ranunculus*, often several metres in length, fill the deeper, mid-channel, areas. Water flow can become very restricted in summer, especially with the encroachment of overhanging riparian vegetation. In late summer, overhanging bank-side vegetation can be such that the effective width of the channel is reduced by more than a metre. Photographs of the Frampton Estate study site are presented in Plate 2.5.



Plate 2.5 a₁ and a₂ Contrasting land use at Frampton Estate: a₁) left bank tilled land with no riparian buffer strip; and b₁) right bank tilled land with buffer strip of complex vegetation.

Plate 2.5 b₁ and b₂ Seasonal changes in vegetation cover at Frampton Estate: b₁) September 2003; and b₂) November 2003

Plate 2.5 c₁ and c₂ Contrasting flows at the Frampton Estate site: c₁) intermediate flow April 2003; and c₂) high flow April 2004.

The chosen discharge cross-section is located in a relatively shallow section in the middle of the RHS reach, adjacent to the middle pressure transducer (see Figure 2.10b). The left bank land use consists of tilled land downstream and parkland upstream and the right bank is also tilled land. On the right bank, both the parkland and tilled land have a five metre wide buffer strip of complex, semi-natural, vegetation beside the river with much vegetation overhanging into the channel.

2.7.1 Data collection and infrastructure at the Frampton Estate site

Fixed infrastructure at the Frampton Estate site consists of three equally spaced pressure transducers (Figure 2.10b). Again, as at Crockways, the pressure transducer stage records are to be used to characterise the water surface slope of the area, which can be used to investigate changes in surface slope in each season. In addition, the middle pressure transducer was used in conjunction with discharge measurements to investigate the changing stage/discharge relationship of the reach. The elevated abundance of vegetation at the site made it an interesting location to develop a stage/discharge rating curve (see Section 2.9.1) as vegetation effects would be more extreme. However, this abundance also made it much less desirable for grid-based measurements; distinctions between open and vegetated areas at this site would be impossible using the 1 m² resolution grid and a finer grid was considered prohibitively time consuming. It was thought that the Hydeclift site would provide a more meaningful contrast to the Crockways site and would be more suitable for the grid-based work. The data collection undertaken at Frampton is described in Table 2.1.

2.8 Research design and methods: macroscale investigations

The macroscale involves investigations which cover spatial ranges greater than 100m and within a timescale ranging from months to years (Carling, 1995). In this project, the macroscale work is used to place the more detailed, smaller scale work in context and provides a temporal and spatial linkage between isolated sites considered at the meso and microscale, and, importantly, also aids the extrapolation of findings for application to other river catchments. This linkage is essential as it enables findings from the present research context to be applied beyond one particular spatial and temporal setting. The macroscale analyses utilised data from the national River Habitat Survey database, bolstered by LOCAR RHS data sets and by primary RHS field surveys and is presented in Chapter 3 of the thesis.

2.8.1 River Habitat Survey

The River Habitat Survey (RHS) was developed in 1994 by the Environment Agency for England and Wales (Raven *et al.* 1998). From 1994 -1996 a national reference network of river sites in England and Wales was established, based on random samples chosen through a systematic stratification of the 10 × 10 km Ordnance Survey National Grid (Jeffers, 1998a). This reference network was extended to cover Scotland and Northern Ireland in 1995. Since then, the RHS database has been regularly supplemented by smaller, more locally focused surveys conducted by the Environment Agency, the Centre for Ecology and Hydrology and various private consultancies. The current database (RHS Version 3.3, 1994 – 2002) holds just over 15,000 entries, to which the 2002 (May – October) baseline surveys of the Frome

and Piddle, commissioned by LOCAR, add a further 177 surveys (109 sites on the River Frome).

The RHS survey provides a simple and standardised method for the assessment of “the character and quality of rivers based on their physical structure” (Raven *et al.* 1998, p. 7). In addition, its use by the Environment Agency includes three further elements: (i) a computer database allowing site comparisons throughout the United Kingdom; (ii) a set of methods for examining river habitat quality; and (iii) a method for evaluating the extent of artificial channel modification (Raven *et al.*, 1998). This project utilises two of these main elements. First, use was made of the national River Habitat Survey database to explore the typicality of the Frome with respect to other chalk river catchments, and, second, the field survey method was used to assess the habitat structure of three detailed study sites on the River Frome.

The RHS field survey sheet is four pages long and incorporates both map based and field derived information. Rivers are assessed in standard lengths of 500m with the main observations based at ten equidistant locations or ‘spot checks’. These spot checks consider channel, bank, and riparian habitat (including up to 50m of the riparian corridor either side of the river). Attributes considered at spot checks include substrate type, aquatic vegetation type and abundance, bank vegetation structure and the type and extent of artificial modifications. In addition, measurements of bankfull width, water width, bankfull height and water depth are carried out at one selected site within the 500 metres, usually centred on a riffle, if present, for consistency and ease of measurement. Finally, a ‘sweep up’ procedure is carried out after all spot checks have been completed; this is to ensure the recording

of infrequent features which may have been missed between spot check locations (Raven *et al.* 1997).

Three sites were chosen for primary field survey on the River Frome. These sections were chosen to provide a backdrop and linkage for the three smaller and more detailed field locations situated at their centre. Raven *et al.* (1997) suggest that the RHS is a useful summary survey and provides an outline of a sites physical character which can then prompt specialist survey. In this study, specialist survey at the reach and micro scale is complemented by the use of RHS. Primary RHS surveys of the three study sites were carried out on four occasions, once in each season, within a one-year study period. This allowed an assessment of the changing habitat structure of the river reaches from winter to summer and provided a more temporally representative context for the mesoscale data sets.

Archival data from both the EA's national River Habitat Survey database and from LOCAR baseline data sets were used to place the field sites and the River Frome in a national context. Site comparisons were made between the chosen field sites and other sites on the Frome, between the Frome and other chalk rivers and between rivers from different geological and geographical areas. Comparisons were made based on physical parameters such as the width/depth ratio and *via* the exploration of in-channel and riparian vegetation characteristics. This analysis was conducted with the subsidiary aim of providing a virtual exploration of, and an introduction to, the River Frome, while the main purpose was to establish whether the Frome might be considered a typical or an extreme example for the study of macrophytic vegetation.

2.9 Research design and methods: mesoscale investigations

The mesoscale, or reach scale, is the next step in the research hierarchy. This involved working at a spatial scale of 10s to 100s of metres and within a time framework of hours or days. The mesoscale is thought to be the most relevant scale for ecological research and is usually typified by work on the scale of the riffle-pool sequence (Carling, 1995). The reach scale is also perhaps the most relevant to river managers, and is the scale most neglected in previous work on macrophyte-flow interactions (see Table 1.3 Chapter 1). Accordingly, the majority of the work for this thesis was carried out at the mesoscale. There are two main foci to the project's mesoscale research: (i) time linked measurements of stage and discharge; and (ii) grid-based measurements of flow velocity, vegetation, and sediment variables. The stage and discharge data are analysed in Chapter 4 of the thesis and the grid-based data are investigated in Chapter 5.

2.9.1 Stage/Discharge Measurements

Given the potential effects of macrophytes on velocity and stage, it was hypothesised that seasonal variations in the stage discharge relationship would occur in approximate sequence with the annual macrophyte growth cycle. This relationship should become apparent from repeated, year-round, time-linked measurements of discharge and stage. Plots of stage against discharge on an annual or multi-annual basis should display differing relationships for the same cross-section according to season. Any observed differences may be attributed to either the effects of vegetation or to scour and fill of the channel bed, and, to isolate vegetation effects,

the extent of scour and fill was determined by using both water depth and stage. The stage/discharge measurements provide a quantifiable measure of vegetation influence on velocity and stage.

The stage/discharge monitoring covered two annual vegetation growth cycles (2003/2004 and 2004/2005) and this was introduced to assess the effects of inter-annual variations in plant biomass. It provides a check not only for the stage/discharge relationship but also for patterns observed in the grid data. Discharge measurements were carried out between September 2003 and September 2005, yielding 100 stage/discharge measurements. Concurrent stage and discharge measurements were taken at two fixed cross-sections at Crockways (GR 612 958) and Frampton (GR 628 947) (see Figure 2.10). Measurements of velocity were conducted using a three-axis Sontek/YSI Flow Tracker handheld Acoustic Doppler Velocimeter (ADV), as illustrated in Plate 2.6. The ADV utilises travelling suspended sediment to track water velocity using an adaptation of the Doppler principle (Sontek, 2002). An underwater acoustic signal of known frequency, or pitch, is generated from the base of the ADV and is reflected back to the probe from particles in the water column towards three inbuilt receivers. Each receiver represents each of the longitudinal, horizontal and vertical flow axes (X, Y, and Z-axes). The ADV is capable of sampling water velocity at 10Hz (10 measurements per second), and can operate in as little as 2cm depth of water. It has a quoted accuracy of “1% of the measured velocity in a one second sample” (Sontek, 2002).

Discharge measurements were taken according to the established USGS/ISO methodology (ISO standards 748 (1979) and 9196 (1992)). The single point method

was chosen and velocity measurements were taken at 0.6 of the depth and over at least 10 verticals in each cross-section. Velocity measurements taken at 0.6 are considered to be a reliable estimate of the average water velocity in a water column and this method was chosen for speed and simplicity. An early check on the efficacy of this was carried out in winter 2003 and showed there was little difference between averaged velocity measurements taken at 0.2 and 0.8 of the depth when compared to a single measurement at 0.6. However, there is some debate as to the validity of this assumption in vegetated channels in the summer season. It is thought that a vegetated bed acts to push the effective average velocity to a higher point in the water column (Gregg and Rose, 1982). This could have potential implications for the present study. In particular, there is a possibility that inaccurate measurement of average velocities in summer could lead to spurious seasonal changes in the observed stage/discharge relationships. However, in this instance, it was thought that to take multiple measures of velocity in the water column would be prohibitively time consuming in a method intended for multiple and rapid repetition. In any case, if the logarithmic profile is disrupted then measurements at 0.2 and 0.8 of water depth would be no more accurate than a single 0.6 depth.

One drawback in using the ADV is that it can suffer from acoustic reflectance from underwater obstacles such as rocks, woody debris and vegetation. Under these circumstances the ADV may record a falsely lower velocity (Sontek, 2002). This problem is particularly pertinent in a study where velocity measurements are required in close proximity to, and often even within, dense stands of vegetation. However, vegetation poses a problem regardless of the chosen measurement probe (Machata-Wenniger and Janauer, 1991). For example, authors in the literature report that

mechanical current meters can become entangled within vegetation (Machata-Wenniger and Janauer, 1991 and Marshall and Westlake, 1990) and that the use of hotwire anemometers is problematic in trailing vegetation, as plant parts may attach to the heated anemometer element (Sand-Jensen, 1996). Other authors, e.g. Eckman, *et al.* (1989), and Petticrew and Kalff (1992), have used pre-weighed blocks of gypsum to measure velocity by changing dissolution rates. These present no vegetation specific problems but the method is laborious and is not refined enough to provide information as to flow direction or variation in velocity over short time scales.

The ADV was chosen with an acknowledgement that a gap was required between the sensor and adjacent objects. The Sontek user manual states that a distance of 15cm is sufficient to avoid interference, and even inside this distance (up to ~ 10cm) the ADV can adapt to and moderate any acoustic reflections (Sontek, 2002). For assurance, a minimum 15cm gap was maintained around the ADV at all times in the field. Within dense stands of vegetation a gap was created just large enough to accommodate this 15cm sampling area: strands of vegetation were held apart by a helper kneeling downstream of the sample location but no plant material was removed. Velocity measurements taken in this way should still accurately reflect their immediate surroundings, especially as flow velocities are generally most affected by objects upstream. As a further check, the ADV automatically records a 'Boundary QC' which describes the effect of any interference on the ADV's performance. This was consulted during data processing to check the validity of results.

River stage was measured continuously by permanent pressure transducer (PT) record using a Druck pressure transducer (Plate 2.7) linked to a Campbell scientific CR10X data logger. The pressure transducers work by monitoring pressure changes exerted by varying volumes of water above them (the influence of atmospheric pressure is removed with reference to measurements taken by a sensor on the river bank). The Campbell data logger converts the pressure difference into a millivolt (mV) signal and this, in turn, can be calibrated in the laboratory to give SI units of length. Laboratory calibrations were carried out prior to field installation by Dr. Joanne Goodson of Kings College London. Six pressure transducers were installed, three at Crockways and three at Frampton (see Figure 2.10), and stage measurements 15-minute intervals. Millivolt measurements were converted to accurate readings of the water level with reference to the individual calibration equations and using the measured height of each pressure transducer unit with reference to a fixed elevation point on the riverbank. Height comparisons between water levels and fixed points were achieved with the use of survey data captured using a Leica TCR 3700 total station. Stage and discharge measurements were then time-matched for analysis.

2.9.2 Stage measurements: data quality control

Some problems were encountered in the running of the pressure transducers which, if undetected, could have introduced spurious trends in the data. The PTs were found to have suffered a periodic change in their measurement baseline, often described as were taken once every second and averaged to give a continuous record of stage at ‘instrument drift’. This measurement error is common with PT measurements (Freeman *et al.* 2004) and was anticipated in this study. Regular checks were made



Plate 2.6 Instrumentation used in stage/discharge measurements: Sontek flowtracker three-axis handheld ADV.



Plate 2.7 Instrumentation used in stage/discharge measurements: Druck pressure transducer and stilling well.

as to the accuracy of the PT readings by comparing stage to manual readings taken from a stage board. Possible reasons for the instrument drift include the age of the probe, changes in barometric pressure, extreme ambient temperatures and leakage of the probe housing (Freeman *et al.* 2004) and, in a chalk river context, gradual calcification of the probes in calcium rich water. Instrument drift was identified and quantified by comparing PT readings to the time-linked manual stage readings and analysing the regression relationships. An example of these manual and automatic comparisons can be observed in Figure 2.11 which shows an instrument drift of ~ 0.08 m for the PT3 probe at Frampton. The recorded drift for the other pressure transducers ranged from 0 cm – 10 cm.

The instrument drift was corrected by applying regression equations obtained from the calibration scatter plots to the PT stage readings. The equations changed depending on the time period of the data and there were a few short periods where the nature of the calibration was uncertain because of gaps in the manual measurement record. Where gaps in the known calibration occurred the values between the ‘known’ periods were adjusted incrementally to produce a smooth and gradual transition between the different regression equations.

2.9.3 Grid-based measurements: vegetation cover and growth form; sediment depth; water velocity; and water depth.

To enable detailed analysis of the mesoscale hydrodynamic effects of vegetation a 20 × 10 m grid was defined across two contrasting river reaches: one shaded and one

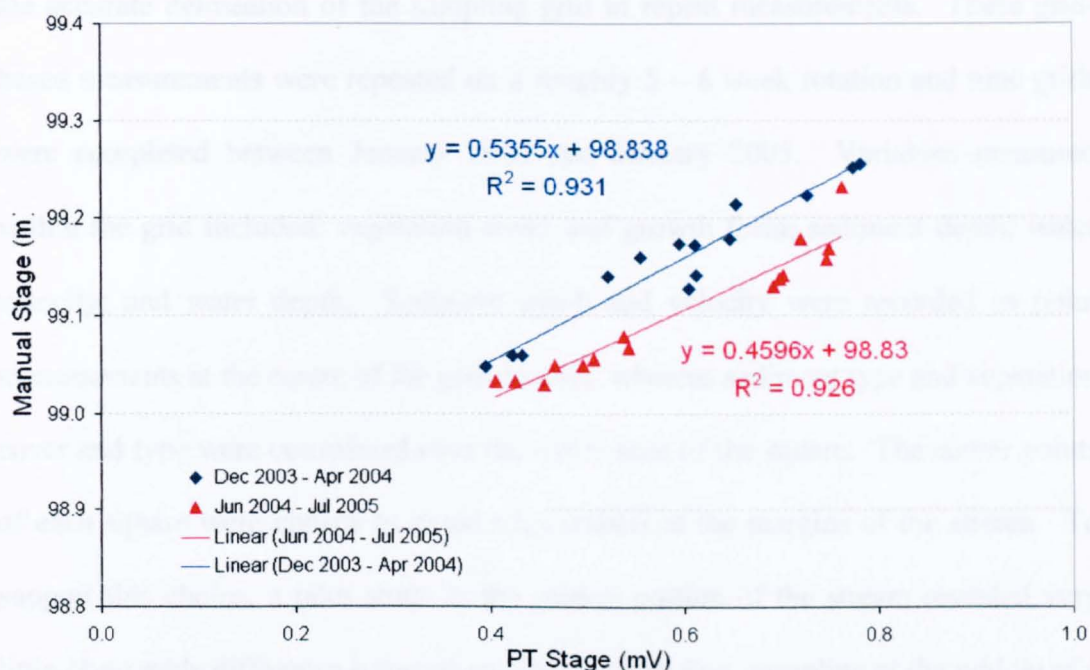


Figure 2.11 Scatter graph and linear regression showing the changing relationship between manual and automatic stage measurements at PT2 at the Frampton Estate field site. Two relationships are evident: (i) December 2003 to April 2004 and (ii) June 2004 to July 2005 and this indicates instrument drift over time. The drift was corrected by recalibrating the raw PT data separately for the two different time periods.

un-shaded. Within these grid limits, measurements were made in each 1m² (see Plate 2.8). This grid-based methodology was chosen so as to operate on a stratified random sampling design and remove sampling bias. While no fixed grid was put in place on the river, markers on the riverbank and the use of retractable tapes allowed the accurate delineation of the sampling grid in repeat measurements. These grid-based measurements were repeated on a roughly 5 – 6 week rotation and nine grids were completed between January 2004 and January 2005. Variables measured within the grid included: vegetation cover and growth form; sediment depth; water velocity; and water depth. Sediment depth and velocity were recorded as point measurements at the centre of the grid squares, whereas sediment type and vegetation cover and type were considered over the entire area of the square. The centre points of each square were chosen to avoid edge effects at the margins of the stream. To support this choice, a pilot study in the central portion of the stream revealed very little observable difference between results obtained from sampling at the middle of a square and sampling at the nodes of the grid squares.

Vegetation cover was estimated by eye as a percentage cover using a list of categories. Traditional methods of measuring vegetation abundance in streams require the removal of large samples of the vegetation to determine either the weight of biomass (e.g. Haslam and Wolsley, 1981) or leaf surface area (Sher-Kaul *et al.*, 1995). A less intrusive method was needed in this study as the repeated grid measurements required vegetation to be undisturbed between site visits. It was decided that an adaptation of terrestrial techniques using visual estimation of percentage cover would be more appropriate. Initially, it had been envisaged that percentage vegetation cover could be better visualised with the use of a subdivided



Plate 2.8 Hypothetical 1m² gridlines superimposed over the Crockways grid site.

quadrat but, after trial use in the field, this proved too cumbersome and time consuming for use in the river environment. It was decided that the use of categorised percentage cover was more appropriate, with acceptance of the attendant loss of detail.

The chosen vegetation recording scale was the Braun-Blanquet Scale. This is the standard scale used in terrestrial vegetation studies (Kent and Coker, 2002), but it has not previously been applied to the river environment. The format of the Braun-Blanquet scale is detailed in Table 2.3. The application of the Braun-Blanquet scale is somewhat qualitative and may be vulnerable to observer bias. With this in mind, care was taken to assign only one person to this task so as to avoid operator variance between surveys. To help visualise the limits of the grid squares, two measuring tapes were strung, 1m apart, across the width of the river while two observers formed the other two sides of each square. The tapes were then moved in metre increments upstream to form each successive 1m wide cross-stream transect. Fine sediment depth was measured using a 'pin-survey' method, which utilised a refinement of Lisle and Hilton's (1992) method for measuring fine sediment depth in pools. A cylindrical metal rod, 2mm in diameter, was used to probe the sediment. Sliding markers were used to first record the level of the water surface on the rod when the tip is resting on the sediment surface, and secondly, to mark the water surface level on the rod when the furthest point of rod penetration has been reached. The difference in position between the two markers is then measured and represents the depth of fine sediment. The position of the sediment surface was determined using visual observations and through changes in resistance transmitted by the metal rod.

Table 2.3 The Braun Blanquet scale used to estimate vegetation cover classes in the grid-based measurements (Modified from Kent and Coker, 2002).

Notation	Braun-Blanquet categorised percentage cover
+	Less than 1% cover
1	1-5% cover
2	6-25% cover
3	26-50% cover
4	51-75% cover
5	76-100% cover

Water velocity within each square was taken as a point measurement at 0.6 of the water depth, the accepted depth for approximating average velocity in the water column. The same limitations discussed in Section 2.9.1, concerning discharge measurements, also apply here, and again this measurement depth was chosen for speed of measurement. Measurements were taken using the 3D Sontek ADV to give a high-resolution measure of velocity in each of the three flow directions. It was anticipated that the assessment of vertical velocities afforded by the ADV would provide measurements with greater relevance to deposition and resuspension processes, whereas previous investigations have only occasionally extended to two-dimensional studies of velocity patterns. ADV velocity measurements were taken at 10Hz and averaged over a period of thirty seconds. The thirty seconds time period was chosen as a compromise between the need to obtain a representative average and important hydraulic constraints, in particular the need to avoid sampling at times of changing stage.

2.10 Research design and methods: microscale investigations

Microscale enquiries were contained within the mesoscale grid reaches but were based at spatial scales of mm or cm and on a time framework of seconds or minutes (Carling, 1995). The main focus of the microscale research was on macrophyte-sediment interactions. Although microscale research has been well represented in previous efforts (see Table 1.3 in Chapter 1), these have focused mainly on velocity-macrophyte interactions and have provided only broad theories and assumptions as regards changes in sediment transport. This study employs new methods and modern equipment to develop a fresh approach to microscale macrophyte-based enquiries

with an emphasis on sediment changes. The microscale analysis is presented in Chapter 6 of the thesis.

2.10.1 Vegetation washing experiments

The initial microscale analyses were mainly laboratory based and focused on the collection of vegetation and sediment samples from the River Frome. For the first part of the investigation, undertaken in October 2003, several samples of aquatic vegetation were taken from the Crockways grid-reach with the intention of examining the nature of any sediment trapped within the plant material and to inform the choice of sediment size used in turbidity experiments (see Section 2.10.2). The aim of this analysis was to determine, for one point in time, the relative amounts of sediment trapped by different plant species, and to quantify the size range of the particles preferentially retained by the vegetation. The October sampling date was chosen to be at the end of the vegetation growth period, when maximum sediment would have accumulated within the plant beds, but before the sediment was washed away by winter high flows (Figure 2.12). It was also important to conduct this experiment outside of the main fieldwork period, as the removal of plant material during the main study period could have greatly affected the mesoscale grid-based measurements (Section 2.9.3). It is acknowledged that the sediment samples represent only one ‘snapshot’ in time, and may have been heavily influenced by antecedent conditions, but it is thought that the samples provide a useful insight into the type of sediment trapped by macrophytic vegetation. Seven vegetation samples were obtained from four plant species, each of different growth form and occupying differing positions in the channel. These included samples of emergent, submergent

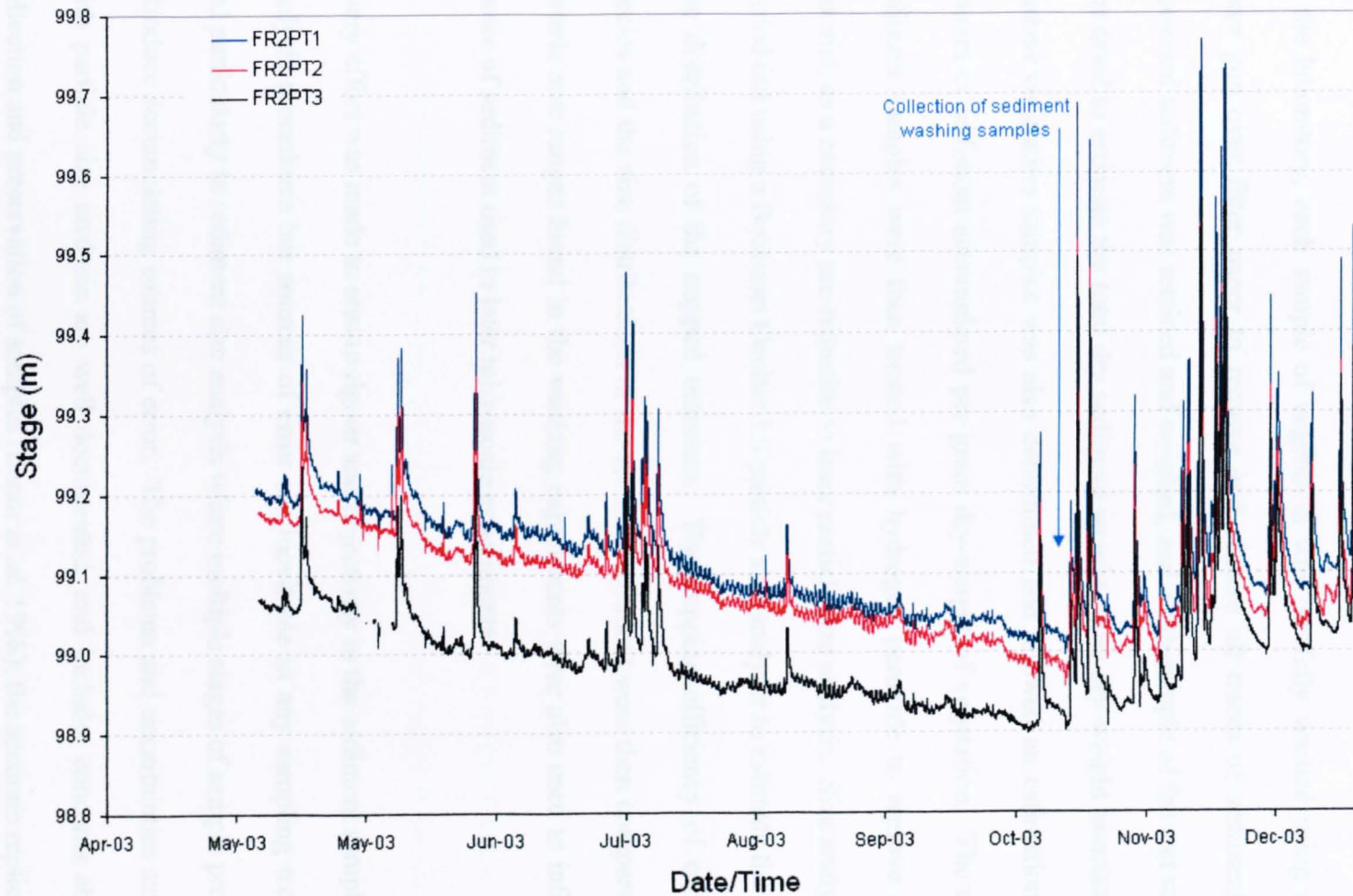


Figure 2.12 Timing of sample collection for vegetation washing experiments at Crockways and the antecedent stage levels for 2003.

and overhanging riparian vegetation. Samples were bagged *in situ*, removed, and taken to the laboratory for analysis.

In the laboratory, each sample of vegetation was carefully washed using a small water gun over filter paper to remove and capture all traces of sediment. The recovered sediment was retained and weighed, and a sub-sample of the wet sediment was dried to estimate the total dry sediment weight. The dry-weight biomass of the washed vegetation samples was also determined and allowed an estimation of the amount of sediment accumulated per gram dry-weight of vegetation. The retained sediment samples were then treated with hydrogen peroxide to remove organic material, as a necessary pre-requisite to laser particle size analysis. Size analysis was carried out using a Beckman Coulter LS particle size analyser to estimate the particle size distribution of the trapped sediment. The trapping efficiency of each plant species and the size distributions of the retained sediment were then compared. The particle size ranges found in the washing experiments were also used to inform the choice of sediment used in later microscale experiments.

Every effort was made to ensure rigour and objectivity in the sediment sampling and analysis procedures but sources of error are inevitable in any sampling technique, and particularly in sediment size analysis where multiple stages of sample processing introduce accumulating sources of error. The problems and uncertainties associated with particle size analysis are well documented, and include: concerns about the collection and preservation of samples (Lenor *et al.* 1998); the accurate replication of the *population* size distribution in sediment *samples* (Swift *et al.*, 1972); the effects of sample pre-treatment procedures (Matthews, 1991a); the accuracy of measurement

apparatus (Swift *et al.* 1972) and factors that complicate size measurement, including aggregate particles (Matthews, 1991a) and differences in particle shape, density and colour (Matthews, 1991b and Clifford *et al.*, 1996). All these potential sources of error and uncertainty are acknowledged here. For example, the use of filter paper was deemed necessary to avoid excessive dilution of the sediment samples. It is accepted that the use of filter paper to capture the sediment may have some implications on particle size; very fine sediments may have been too fine to be retained by the filter or may have been trapped within the matrix of the filter paper. The filter paper was chosen to provide a balance between the need to retain fine particles and adequate filtration speed and paper strength. Accordingly, Whatman 1 filter paper was selected. This has a pore size of 11 μ m and most particles below 11 μ m will thus have been lost through the filter. However, 11 μ m is within the 'fine silt' category on the Wentworth scale, meaning that only fine to very fine silt and clay may have been lost. These size classes are generally very difficult to remove from suspension but may be attracted to the plant by electrochemical processes. Cotton *et al.* (2006) suggested that less than 10% of the fine sediment accumulations found below *Ranunculus* in the River Frome were of the silt and clay fraction (0.37 - 63 μ m). All the size distribution graphs show an expected low response below 11 μ m (see for example Figure 2.13). Some response is evident below 11 μ m and this suggests that some smaller particles were present, due to either clogging of the paper filter or disaggregation of the original particles during pre-treatment.

The samples underwent two pre-treatment procedures before use in the size analyser which may have altered the sediment composition. Organic material was removed

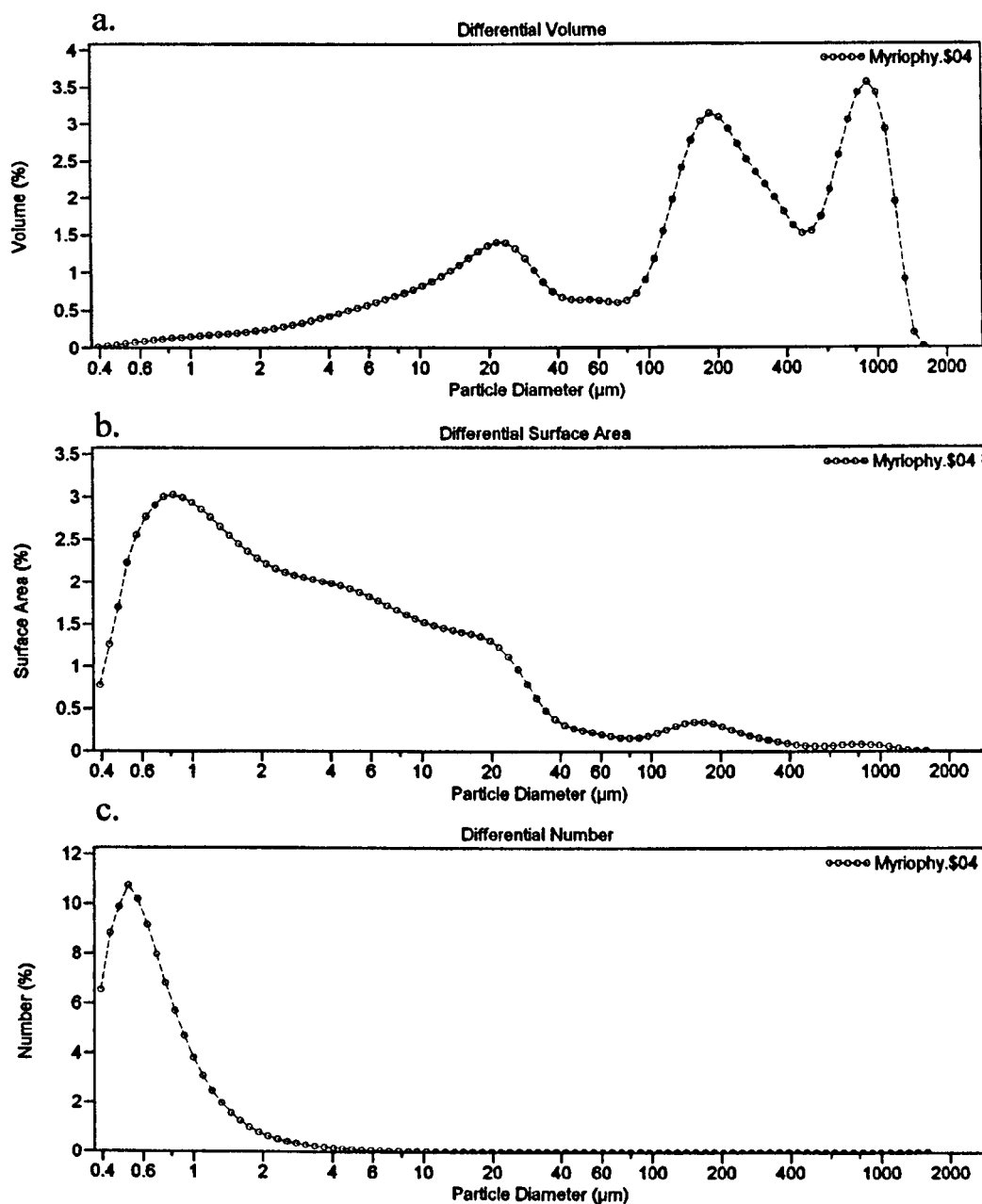


Figure 2.13 Particle size distribution, displayed by (a) volume (b) surface area and (c) number of particles, for a sediment washings sampled from *Myriophyllum alterniflorum*.

using hydrogen peroxide to avoid fouling of the size analyser, and mineral particles greater than 2 mm were removed by wet sieving to prevent blockage of the machine.

The removal of organic matter eliminates a potentially important component of the sediment load found in the plant washings (Wharton *et al.*, 2006) and also causes disaggregation of mineral particles (Matthews, 1991a), meaning smaller particle sizes than the reality may have been recorded. However, this experiment was principally designed to determine the size of material to be used in later turbidity experiments (Section 2.10.2) which utilised mineral and not organic sediment and is in accordance with the methods of Cotton *et al.* 2006 who investigated the particle size distribution of sediment trapped beneath macrophyte stands. Cotton *et al.* did determine the proportion of organic sediment in their samples by loss on ignition. In this study the method of particle size analysis required that samples remained wet and measurement of dry-weight organic matter through loss on ignition was not possible on the whole sample. The sediment samples obtained in the washings were too small to allow a sub-sample to be tested. The removal of mineral matter greater than 2 mm was less influential; only two samples, both from the submerged macrophyte *Myriophyllum alterniflorum*, were found to have particles over 2mm. These outsize particles were weighed and were found to be relatively insignificant, comprising only 5.5 % and 0.2 % of the total samples by weight.

Finally, a volume-based sediment distribution for the sample was chosen and this itself may cause uncertainty in interpretation. Volume-based measurements can be misleading because a small number of large particles may have the same volumetric influence as a large number of small particles. Conversely, however, the use of

count or surface area measurements will maximise the influence of small particles and conceal the presence of larger particles (Figure 2.13). Volumetric display was chosen to best display diversity in the samples. Fractional weight was not used to display the data because the density of the sediment particles in the mixed sample was unknown.

2.10.2 Microscale turbidity measurements

The principal focus at the microscale was on a series of experiments aimed at capturing the influence of vegetation on sediment in transport. In these experiments five IR40C turbidity probes were placed upstream, downstream, within and alongside vegetation stands, to record the influence of the vegetated filter on the passage of an introduced sediment signal. It was anticipated that the high-periodicity turbidity probes would discern modifications made by the vegetation to either the sediment concentration or the speed of travel of the sediment pulses. The passage of suspended sediment pulses through a vegetated patch was compared to an unvegetated control experiment to help account for any dispersion effects. The turbidity experiments were carried out in July 2005, after the main field monitoring had ceased. This period was chosen for three reasons: (i) to ensure that the artificial release of sediment would have no effect on grid-based sediment measurements; (ii) to guarantee non-flood conditions – the turbidity experiments would not have been possible during storm flows of high background turbidity; and (iii) to provide a wide range of vegetation patch sizes. An example of an artificial sediment release is illustrated in Plate 2.9.



Plate 2.9 Artificial sediment release at Hydeclift Plantation study site, July 2005.

Variety in vegetation patch size was essential in providing a seasonal analogue to help account for changes in plant-sediment interactions induced by temporal changes in biomass. The July sampling period coincided with high plant biomass and diversity in the channel and allowed adequate 'space for time substitution'. Diversity of plant growth-form was also considered important, and initially it was hoped that both the dominant submergent (*Ranunculus calcareous*) and emergent (*Sparganium erectum*) species could be utilised in the analysis. However, given the great length of the *Sparganium* stands (> 10 m), and the low velocities associated with their marginal location, it was decided that the turbidity experiments were only feasible for mid-channel, *Ranunculus* stands which were generally less than 7.5 m in length and experience higher velocity flows. A larger number of turbidity probes may have allowed both emergent and submergent vegetation to be considered but this would have greatly complicated both the analysis and interpretation of the data. Three ranked sizes of *Ranunculus* stands were chosen: small (~ 3 m), medium (~ 4.5 m) and large (~ 7 m) and these represent the dominant vegetation stand sizes expected in spring, early summer and late summer respectively. Space for time substitution was also exploited to help illustrate changes in sedimentation processes with changes in water velocity and depth. Similar turbidity experiments were carried out at both the Crockways and at Hydeclift grid-reaches to provide an analogue for changing depth and velocity conditions. As described in Chapter 6, there is a marked contrast in physical and hydraulic conditions between the two grid-reaches; Hydeclift is highly uniform with shallow water and high velocities, whereas, Crockways is deeper, has more varied in-channel features and is of comparatively lower velocity, especially in summer.

Four different types of sediment were used in the turbidity experiments. Three were sieved in the laboratory to obtain precise size ranges of 0-1 phi (1000 – 500 μm), 1-2 phi (500 – 250 μm) and 2-3 phi (250 – 125 μm). These correspond to ‘coarse’, ‘medium’ and ‘fine’ sand particles on the Wentworth scale. The fourth sediment type was collected from a marginal sediment bank at the Hydeclyft field site and was used untreated in the turbidity experiments. A sample of this sediment was retained and later used to determine the size distribution of the field sediment. It was found to have a mean particle size (by volume) of 105 μm (3 - 4 phi) corresponding to ‘very fine’ sand on the Wentworth scale. Collectively, these four sediment types cover the dominant sediment size range obtained for submerged macrophytes in the vegetation washing analyses (see Figure 2.14). The use of different sediment size ranges allowed investigation of the effects of the vegetation on different particle sizes in an attempt to define the size of the ‘vegetative filter’. A comparison of the turbidity traces of the different size classes should show which particle size is most affected by the vegetation. The use of size-restricted sediment tracers was also beneficial to reduce the uncertainty of turbidity measurements. The response of turbidity probes such as the IR40C are sensitive to particle size effects and the calibration relationship will differ markedly for different particle sizes (Clifford *et al.*, 1995). Mixed sediment, i.e. samples consisting of widely varying sediment sizes, will create a stable response from the turbidity probes only if the same proportions of the mixed sediments remain in suspension at all times. With the microscale experiments it was theorised that the vegetation would preferentially retain certain particle sizes and that this would invalidate this assumption and it was necessary that sediment with a narrow size distribution be used. The turbidity probes were calibrated separately for

each sediment type using representative samples in the laboratory and following the method of Clifford *et al.* (1996) see Section 2.10.3.

Probes were deployed in a 'probe array' with five Partech IR40C turbidity probes and a velocity meter operating concurrently and taking high frequency measurements at a rate of 5Hz (Figure 2.15). Trials of the equipment carried out at a higher sampling rate of 10Hz showed that measurements became unreliable; large chunks of data were lost as the CR10X logger became overloaded by the data supplied from multiple sensors at such high a frequency. Velocity measurements at the microscale were taken using a Valeport 2D electromagnetic current metre (EMCM). This was modified by the manufacturer to permit compatibility with a Campbell CR10X data logger. This allowed measurements to be taken at 5Hz and ensured that velocity measurements could be easily time-linked to the turbidity measurements.

2.10.3 Calibration of IR40C turbidity metres

Measurement of suspended solids was conducted using Partech IR40C infra-red turbidity probes to obtain surrogate values of suspended sediment concentration. These instruments transform the light attenuated by passing sediment particles into a voltage output (Clifford *et al.*, 1995) which serves as an analogue for suspended sediment concentration. These high-resolution instruments are most often deployed in long term stationary positions, though Clifford *et al.* (1995a, 1995b and 1996) have previously used them in a more dynamic role. This research utilises this more active approach and applies it to the study of macrophyte-sediment interactions to

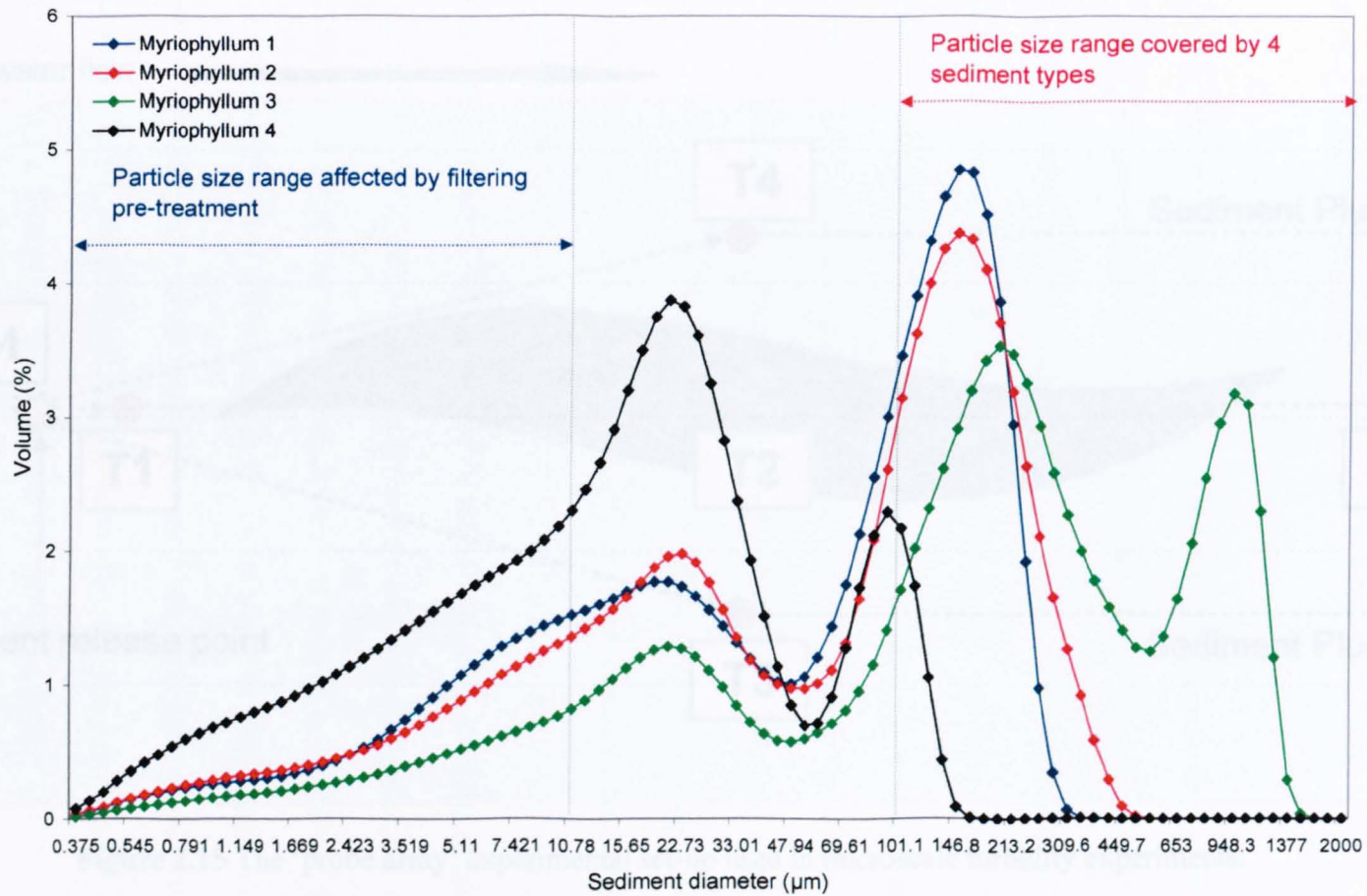


Figure 2.14 The collective range of the four sediment types used in the turbidity experiments and their relation to the dominant sediment size range obtained for submerged macrophytes in the vegetation washing analyses.

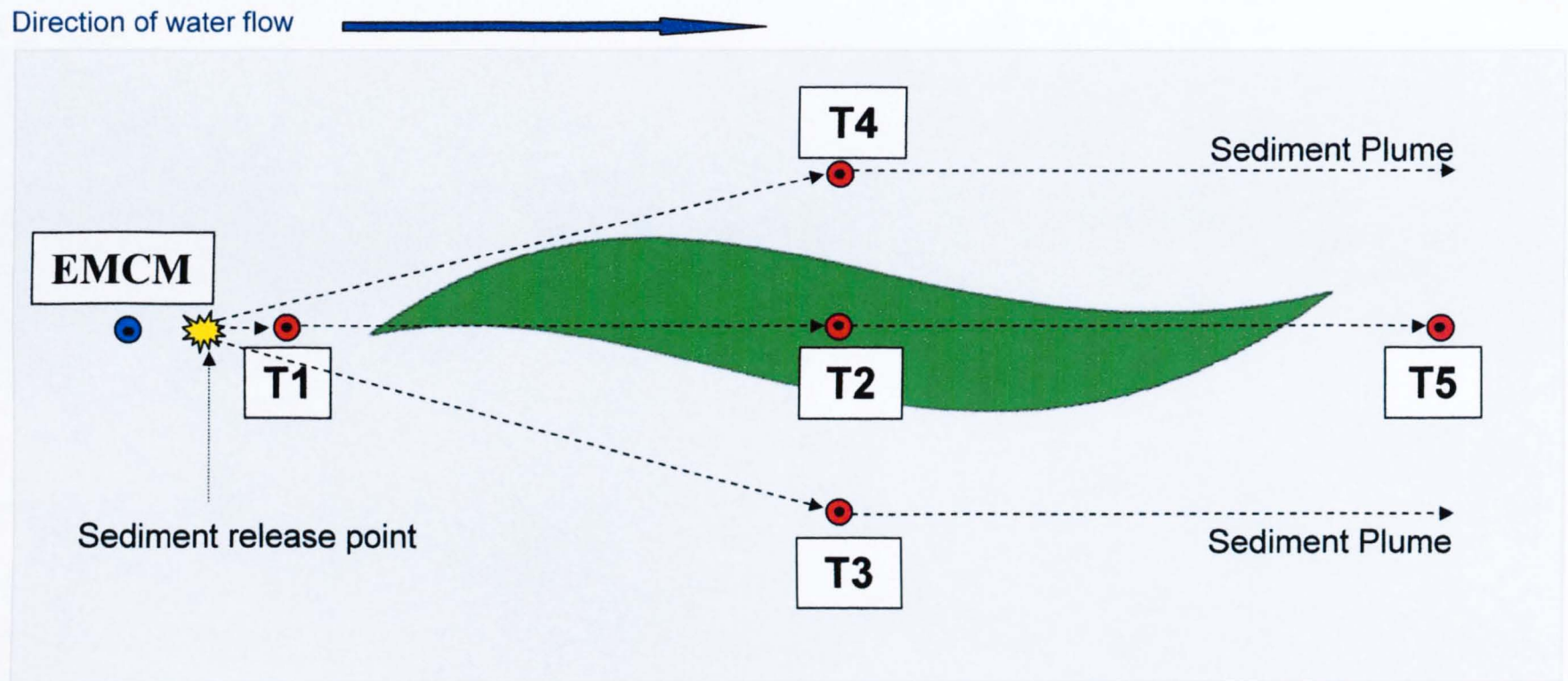


Figure 2.15 The 'probe array' experimental set-up used in microscale turbidity experiments.

gain an insight into small-scale suspended sediment transport processes and the modifications introduced by macrophytic vegetation. Measurements in the field were taken with the IR40C connected to a Campbell scientific CR10X data logger and using a purpose-written program.

The turbidity probe response may be influenced by changes in particle size, shape, density and colour (Matthews, 1991b; Clifford *et al.* 1996) and the voltage readings must be converted to a more meaningful measure of total suspended sediment flux by calibration with known sediment quantities in the laboratory (see Figure 2.16). During calibration, the turbidity probes were placed in a large, dark coloured container filled with a known quantity of water, to which successive additions of known quantities of sediment were added. This water-sediment mix was agitated mechanically to keep the particles in suspension and the average millivolt (mV) response of the probes at each sediment concentration was recorded. Turbidity measurements were taken at a rate of 5Hz and averaged over 30 seconds to give a representative result. The mV values of turbidity measured by the sensors were then compared to the corresponding sediment concentrations to develop a calibration curve. This process was repeated separately for each of the sediment types used in the field experiments. However, for the largest, and heaviest, grain size used (0-1 phi sand), calibration was not possible as the grain sizes proved too heavy to keep in suspension. The resulting calibration curves for the marginal field sediment (3- 4 phi), 2-3 phi sand, and 1-2 phi sand are shown in Figure 2.16.

2.11 Discussion

This chapter has outlined the physical characteristics of the River Frame and the site selection process used in this study. The main field study sites were chosen through a combination of stepwise elimination of less suitable sites and purposive sampling based on the requirements imposed by the thesis methods. It was anticipated that the

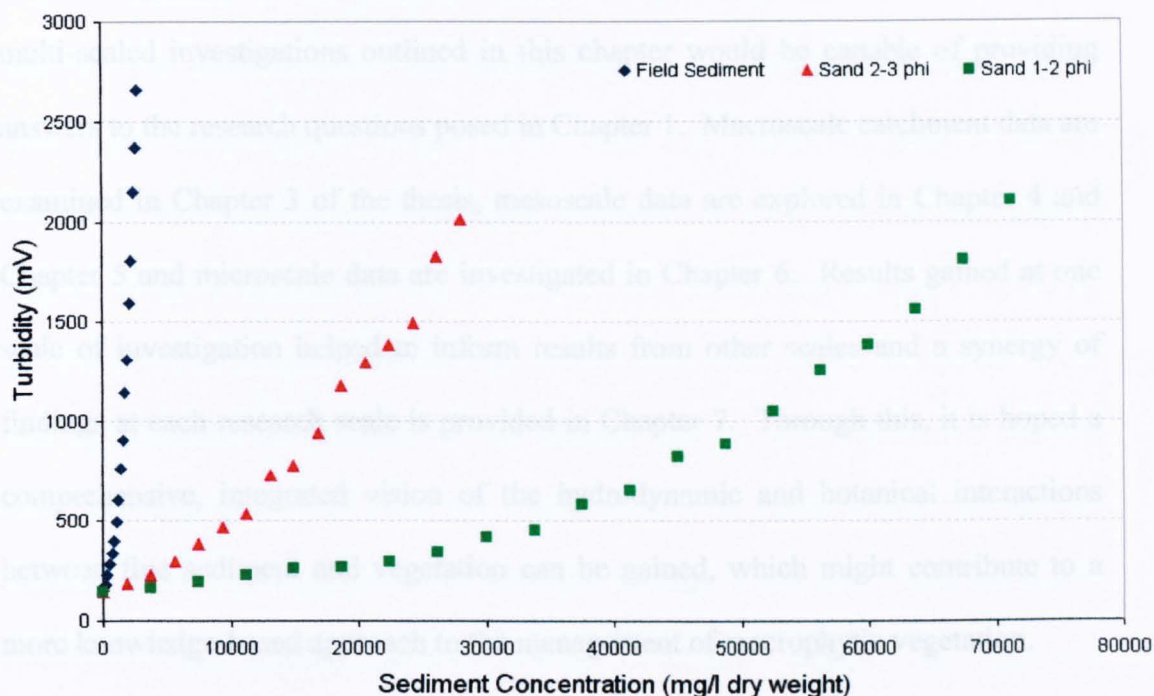


Figure 2.16 Calibration curves for three of the sediment types used in microscale turbidity experiments. The influence of particle size on the turbidity probe response is easily apparent: the fine sediment exhibits a higher turbidity response than the coarse sediment at an equivalent sediment concentration.

2.11 Discussion

This chapter has outlined the physical characteristics of the River Frome and the site selection process used in this study. The main field study sites were chosen through a combination of stepwise elimination of less suitable sites and purposive sampling based on the requirements imposed by the thesis methods. It was anticipated that the multi-scaled investigations outlined in this chapter would be capable of providing answers to the research questions posed in Chapter 1. Macroscale catchment data are examined in Chapter 3 of the thesis, mesoscale data are explored in Chapter 4 and Chapter 5 and microscale data are investigated in Chapter 6. Results gained at one scale of investigation helped to inform results from other scales and a synergy of findings at each research scale is provided in Chapter 7. Through this, it is hoped a comprehensive, integrated vision of the hydrodynamic and botanical interactions between fine sediment and vegetation can be gained, which might contribute to a more knowledge-based approach to the management of macrophytic vegetation.

3. THE CATCHMENT IN CONTEXT: RIVER HABITAT SURVEY

3.1 Chapter synopsis

The general characteristics of chalk rivers, according to the published literature, have been summarised in the previous chapter to give a predominantly qualitative description of chalk streams. The literature champions the idea of a distinct chalk ‘river type’ which can be defined as in Table 3.1. However, these suggested definitions have been almost exclusively informed by (i) qualitative observations from a number of chalk rivers (e.g. Westlake, 1972; Mantle and Mantle, 1992) or by quantitative observations from only one river (e.g. Dawson, 1976). In contrast to the existing literature, the analyses in this chapter aims to use quantitative analysis examine a large number of chalk rivers from each of the different geographical and climatic areas across England and to relate the ‘average’ chalk river characteristics to rivers of different geology across the UK and to the River Frome. The analysis in this chapter aims to test the validity of the general consensus that chalk streams form a discrete river sub-set, and to assess the accuracy of the literature in defining the chalk river type. Comparisons are also made between the chosen study sites and the average indicators for the River Frome. The national River Habitat Survey database and LOCAR RHS database were used to define quantitatively five aspects of river sites of chalk geology which have relevance to the study of in-channel vegetation: energy regime; channel dimensions; vegetation biomass; riparian tree cover; and substrate type. The methods of RHS field survey have been described in Section 2.8.1 of Chapter 2.

Table 3.1 Summary descriptors of the chalk river group and the River Frome with reference to previous research outlined in Chapter 2.

Descriptor	The chalk group	The River Frome
Discharge regime	Stable river flows; major flooding rare; predictable seasonal discharge pattern: min flow in summer, max in winter. Max to min flow of 10:1.	Stable river flows; major flooding rare; predictable seasonal discharge pattern: min flows in summer, max in winter.
Fauna	Diverse and productive communities of high biomass; each plant species supports a specific faunal assemblage.	Diverse and productive communities of high biomass; each plant species supports a specific faunal assemblage.
Flora	Summer plant biomass may reach 400 g dry weight m ⁻² , but typically 200g dry weight m ⁻² ; longitudinal change in plant species type from source to mouth.	Average biomass 200g dry weight m ⁻² ; ranging from 30g dry weight m ⁻² in deep water to 300 – 400 g dry weight m ⁻² in shallow water; longitudinal change in plant species type from source to mouth.
Human impacts	Clearance of riparian tree cover; water meadows and supply channels; commercial watercress farms and fishing; vegetation cutting and removal for flood prevention; groundwater and surface water abstraction.	Clearance of riparian tree cover; water meadows and supply channels; commercial watercress farms and fishing; vegetation cutting and removal for flood prevention; groundwater and surface water abstraction.
Substrate and sediment	Coarse gravel and cobble beds with flints and concretion; discrete deposits of fine, organic sediment beneath plant beds.	Coarse gravel and cobble beds with flints and concretion; discrete deposits of fine, organic sediment beneath plant beds.
Water quality	Low turbidity; high nutrient availability; pH 7.4 – 8.5; maximum seasonal temperature range of 2 – 25°C and monthly mean variation < 8°C; 83 % of chalk rivers of good chemical quality (2000).	High levels of phosphate and nitrate; pH 7.4 – 8.4; temperature ranges from 5.5 – 7.5°C in January to 16.0 – 18.5°C in August. Range of annual means (1969 – 1972) of 10.4 – 10.8°C.

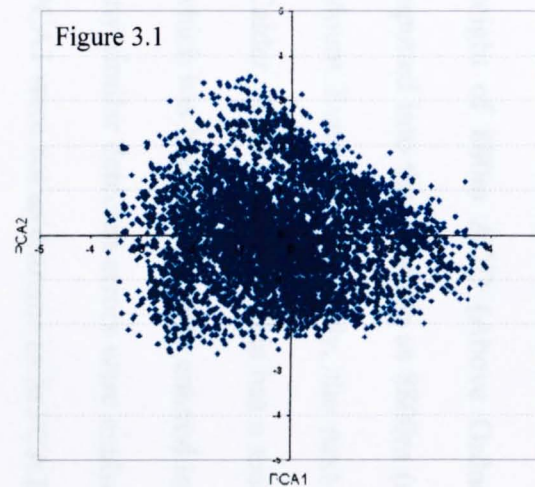
3.2 Energy regime: a typology of UK rivers

One of the Environment Agency's four original aims for the RHS initiative was to use the database to develop a statistical typology, or classification, for all rivers in the United Kingdom (Raven *et al.*, 1998). After initial examination of the database, it was discovered that many field-derived habitat variables, for example substrate type, were highly correlated with five map-based variables: altitude; slope; distance from source; height of source and solid geology (Jeffers, 1998b). However, these variables were also highly correlated with each other, and, to remove any covariance, the first four of these variables were placed in a principal component analysis (geology was not included because it is a categorical parameter). This linear transformation reduced the original four correlated variables into two new uncorrelated and orthogonal principal components: 'PCA1' and 'PCA2' (Jeffers, 1998b). PCA1 represents the 'local profile' of a site and combines measures of altitude and slope, it accounts for 54.1% of the variability measured by the original four variables. PCA2 is an analogue for 'potential energy' and combines height of source and distance from source (Jeffers, 1998b). PCA2 accounts for 32.3% of the variability measured by the original four variables. Together, both variables retain 86.4% of the variability in the original data set.

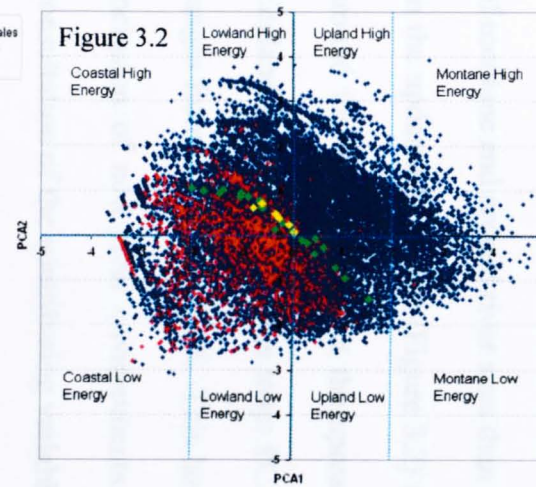
PCA scores are available for each site in the RHS database and these can be used to plot a PCA plot: this is essentially a two-axis scatter graph of the two principal component scores for each river site. Jeffers (1998a) presented a PCA plot for the 4569 sites in the baseline England and Wales database (1994 – 1996) and this is reproduced here in Figure 3.1. In Figure 3.2 Jeffers' (1998a) original PCA plot has

been updated for this thesis to include all viable entries from the circa 15,000 sites in the present RHS database (v3.3, 1994 - 2002). The updated PCA plot is also subdivided to illustrate the various mapped positions of: (i) chalk streams; (ii) chalk sites on the River Frome; and (iii) all sites on the Frome regardless of geology. In addition, Figure 3.3 shows the Frome RHS database sites in more detail and Figure 3.4 provides a comparison of the 2002 LOCAR RHS data collected for the Rivers Frome and Piddle.

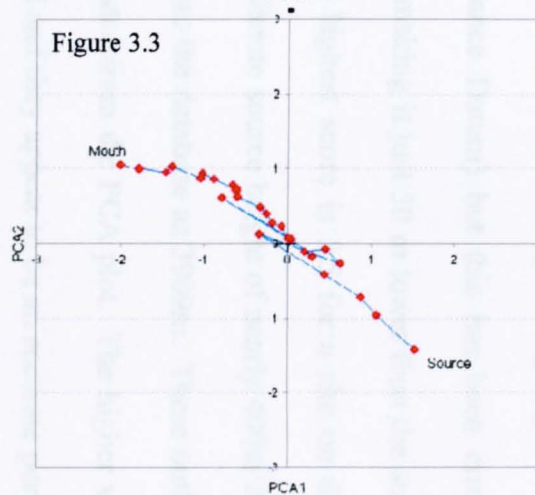
The two axes of the PCA plot are considered to have environmental significance. Both axes reflect their original component variables and form environmental gradients between: (i) coastal, low altitude, environments to high altitude, montane environments (PCA1); and (ii) low energy to high energy environments (PCA2). These continua have been arbitrarily divided to obtain eight river types (see Figure 3.2). The arbitrary divisions denoted in Figure 3.2 delineate the boundaries of each river type but it is the points at the 'centre of gravity' of a particular river type that define its character (Environment Agency, 2000). With increasing distance from the centre of gravity sites become less and less representative and comparisons become less and less valid (Environment Agency, 2000). Some points may even lie on the boundary between two or more river types and the classification of a single river can change significantly from source to mouth. Given the arbitrary nature of these divisions, it is often more useful to look at the position of a site in the overall two-dimensional ordination rather than the general river type. This is a much better guide to the probable habitat features present at a site than the arbitrary classification (Jeffers, 1998b).



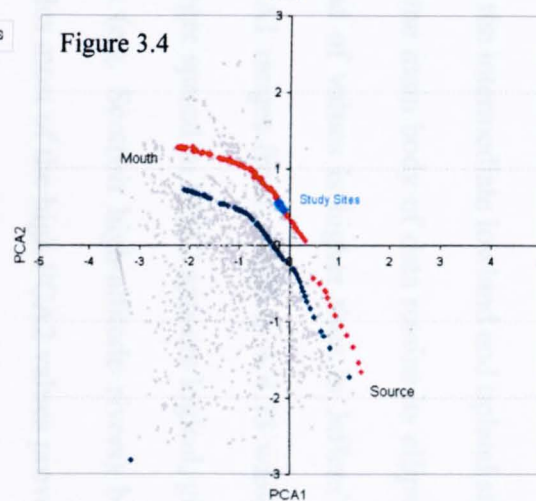
• England & Wales
baseline sites
(1994 - 1996)



• All other sites
• All other Chalk
• All other Frome
• Frome chalk



• Frome RHS



• Chalk sites
• Piddle LOCAR
• Frome LOCAR
• Study Sites

Figure 3.1 Jeffer's PCA plot of baseline RHS sites in England and Wales (1994 – 1996).

Figure 3.2 Updated PCA plot containing c. 15000 UK sites from the RHS database version 3.3 (1994 – 2002) subdivided by site geology.

Figure 3.3 Line plot of PCA scores for the Frome indicating progressive changes in PCA scores with increasing distance downstream.

Figure 3.4 PCA plot of sites for the Frome and Piddle from the LOCAR database set against a background of all chalk sites.

3.2.1 Comparison of the baseline and the updated PCA plot

Jeffers (1998b) describes the plot of data in the baseline PCA plot as forming a “rough ellipse” around the axes origin (Jeffers, 1998b p. 532). The spread of values is slightly larger on the PCA1-axis than on the PCA2-axis; the values of PCA1 range from - 3.9 to + 4.1, while PCA2 ranges from – 2.82 to + 3.37. The data coverage is less dense at the top left and bottom right of the ellipse, indicating a smaller number of montane and coastal river sites than the intermediate lowland and upland sites.

In the updated PCA plot (Figure 3.2) the main body of data retains its ellipse shape around the axes origin, but the spread of values is higher than in Jeffers’ (1998) England and Wales baseline map: PCA1 ranges from – 3.68 to + 4.73 while PCA2 ranges from – 3.10 to + 8.56. This larger spread in PCA2 appears logical, given the inclusion of new river environments (e.g. Scottish high-altitude rivers) but after consultation of the contributing variables most of the high PCA2 values proved to be erroneous. For example, the upper limit of the range for PCA2 (8.56) is an outlier value for the River Teith in Scotland. The Teith has a relatively high true source height of 880m AOD (Above Ordnance Datum) but this has been erroneously inputted into the database as 8800m (making it just 50 m lower than the summit of Mount Everest). Similarly, the next highest score is 5.5 for a site on the River Calder, West Yorkshire; this has a moderate source height of nearly 400m AOD but which has been erroneously entered into the database as 3909m. These outliers and any similar detected errors were excluded from the PCA plot. The higher values of PCA1 were not as extreme as in PCA2 and they appear to be an accurate portrayal of the higher altitude and slope of Scottish sites. For example, the highest PCA1 score, 4.73 is for a site on the Coire Etchachan Burn, in the Cairngorms, which has both a

high altitude of 897 m and a steep slope of 133 m/km. After the exclusion of false values, the new range for PCA2 was – 3.1 to + 3.92.

3.2.2 Chalk rivers, the Frome and chalk sites on the Frome

The updated PCA plot has been divided into four sub-groups: (i) Chalk sites on the River Frome; (ii) Frome sites of all other geology; (iii) all sites of chalk geology; and (iv) all remaining sites in the database. Sites of ‘chalk geology’ were extracted from the database using a single criterion relating to solid geology and using the RHS category of ‘chalk, including red chalk’. It is acknowledged that this method of selection excludes individual sites on chalk streams that occur on other rock types but which are heavily influenced by the predominance of chalk rock in their catchment. Other selection criteria were considered including ‘river name’, ‘LEAP/catchment name’ and ‘county name’. However, no comprehensive list of river names exists for the UK (Jeffers, 1998a) and more importantly, delineating which portions of recognised ‘chalk streams’ or ‘chalk areas’ should be included, and which should be excluded, would have been entirely subjective. It was thought that only including river sites directly underlain by chalk rock would provide a more objective and representative list of chalk river sites. To conform to the exclusion of chalk-influenced sites in the main chalk data body, the sites on the River Frome were also divided into sites of chalk and non-chalk geology and treated separately.

In the subdivided PCA plot the general data-body shape necessarily remains the same, but the different coloured groupings plot in specific areas of the ordination. The first division to notice is the difference in plot location for the chalk and non-

chalk geology groups. The 'other geology' sites generally plot as the total data set plots and seem relatively unaffected by the removal of several data subsets. The chalk group, however, forms a distinctive cluster to the bottom left of the main data body. A clear presence and absence is evident in the distribution of chalk sites: the chalk group does not appear in either of the montane river types and is virtually absent from the upland high-energy river type. Chalk rivers are most highly represented in the lowland low-energy and lowland high-energy categories, with several, scattered, points in the coastal and upland low-energy classes. The location of the chalk sites is not unexpected given the environmental gradients represented by the two principal components; chalk rivers originate on low altitude hills of moderate slope and this is reflected in the position of chalk sites in the data plot. Table 3.2 shows the comparative baseline descriptive statistics for slope, altitude, distance from source and height of source for chalk rivers compared to all other rivers in England and Wales, Scotland and Northern Ireland. Chalk sites score consistently lower than the UK average in all four criteria.

The second point of interest is the location and orientation of data points representing the River Frome. Both 'chalk' and 'non-chalk' Frome sites plot along the top boundary of the chalk group and almost form a 'boundary line' at the edge of the chalk data body. As the two-axes of the map have an environmental significance, then an explanation of the Frome's position should lie within the original four variables. Table 3.2 contains only five Frome sites but this is enough to demonstrate that the River Frome has a slightly higher altitude, slope and source height than the chalk average. In particular, the source height for the Frome is very high, at more than twice the height of the chalk average. The other interesting aspect of the Frome

Table 3.2 Descriptive statistics for four correlated map-based variables for six subsets of the RHS baseline database (1994 – 1996). These four variables can be combined to produce two uncorrelated principle components: PCA1 and PCA2 which may be used produce a statistical typology of UK rivers. Adapted from Jeffers, 1998b.

Variable	Sample Number	Mean	Median	Maximum	Standard Deviation
England & Wales 1994 - 1996					
Altitude	4569	79.9	55.0	630	86.1
Slope	4569	8.45	3.3	200	16.5
Distance	4569	18.5	8.0	263	28.5
Height	4569	217.4	147.0	800	184.0
Scotland 1995 – 1996					
Altitude	779	129.9	90.0	645	117.3
Slope	779	25.0	13.3	300	33.1
Distance	779	12.2	6.5	160	18.3
Height	779	425.3	400.0	1225	257.9
Northern Ireland 1995 - 1996					
Altitude	267	70.8	58.0	280	52.3
Slope	267	7.93	3.3	100	12
Distance	267	18.4	12.0	141	19.7
Height	267	283	250.0	610	134.4
UK 1994 - 1996					
Altitude	5615	86.9	55.0	650	91.5
Slope	5615	11.3	3.3	300	21.0
Distance	5615	16.5	7.0	263	25.9
Height	5615	248.4	145.0	1225	208.2
Chalk Rivers 1994 – 1996					
Altitude	514	45.5	37.5	185.0	36.8
Slope	514	2.6	1.7	84.5	2.7
Distance	514	15.3	8.0	178.0	23.4
Height	514	82.7	84.5	230.0	44.9
River Frome 1994 - 1996					
Altitude	5	58.0	45.0	130.0	49.6
Slope	5	3.8	2.6	9.1	2.9
Distance	5	22.5	26.0	39.5	15.8
Height	5	170	170	170	0
River Piddle 1994 - 1996					
Altitude	5	54.6	50.0	105.0	38.2
Slope	5	4.4	3.3	10.0	3.5
Distance	5	14.0	11.0	27.5	10.9
Height	5	130	130	130	0

plot is its shape and orientation; the data points form a thin, curvilinear trace which crosses through the map origin and is orientated bottom-right to top-left, from the upland low-energy river type to the lowland high-energy. A line plot of the Frome's PCA scores arranged according to their longitudinal position (Figure 3.3) reveals that, in general, the PCA scores change progressively with distance downstream, forming an approximate continuum from highland to lowland. This is entirely predictable, as altitude and distance from source will necessarily change with increasing distance downstream.

The chalk sites plot almost perfectly onto the non-chalk sites, suggesting that there is little difference in the energy regime experienced at chalk and non-chalk sites on the Frome. The chalk sites cluster within the middle of the Frome sites and this suggests that chalk rock is most dominant in the central portion of the river. The geological map of the Frome (Figure 2.2) in Chapter 2 confirms this.

3.2.3 The Frome versus Piddle

Figure 3.4 shows the PCA scores from the LOCAR RHS surveys of the River Frome and the adjacent River Piddle within the background context of all chalk streams. The longitudinal transitions are perhaps more evident in this plot than in Figure 3.3 given the more complete survey coverage for both rivers, but a few discontinuities in the longitudinal ordering of sites exists. The Frome has consistently higher PCA scores than the Piddle, which plots closer to the centre of the main chalk data body than the Frome. This is again a consequence of the four contributing variables: table

3.2 shows that the Frome has a higher source height and average site altitude than the River Piddle.

3.2.4 Energy regime at the River Frome study sites

The LOCAR RHS sites closest to the chosen study sites have also been distinguished on Figure 3.4. These sites are located within the central portion of the plot line and represent sites underlain by chalk rock and of high source height (189 m), a moderate distance from source (average 16.1 km), low altitude (average 79.2 m) and a relatively high slope (average 2.73). The sites correspond to ‘lowland high energy’, based on the arbitrary EA classification. Table 3.3 details the PCA scores, source height, distance from source, altitude and slope for each of the study sites on the River Frome. The PCA1 and PCA2 values for the study sites were obtained by matching the grid reference values for the LOCAR RHS sites to the grid references for the study sites. The PCA values demonstrate that the three study sites on the Frome have similar energy regimes and this is also demonstrated in Figure 3.4, where the study sites appear on the PCA plot as lozenges within the red trace of RHS sites on the Frome. The three study sites plot in the centre of the Frome values and can be described under the Environment Agency typology as ‘lowland high energy’.

3.3 Width depth ratio

The ratio of width to depth is a classic method of measuring channel shape and is found by dividing the width of a channel cross-section by the depth. In the RHS, width and depth are recorded at one representative cross-section within the 500m

Table 3.3 Descriptor values for the three study sites on the River Frome. Width depth ratio, total tree score, summer ICV index and substrate index were obtained through primary RHS surveys, while energy regime values were obtained from corresponding sites in the LOCAR RHS database.

Study Site	Energy Regime		Width Depth ratio	Total tree score	Summer ICV index	Substrate index
	PCA1	PCA2				
Crockways	-0.12	0.46	11.6	5	1.65	-1.1
Hydeclift Plantation	-0.18	0.52	10.7	8	1.8	-5.1
Frampton Estate	-0.21	0.55	19	6	4.85	-2.8

reach, usually centred on a riffle for consistency and ease of measurement. River dimensions are recorded as: (i) bankfull width (the horizontal distance across the channel at the level where the river first spills out onto the floodplain); and (ii) banktop height (the vertical distance from the water surface to the first major break in slope after which cultivation or development is possible (RHS manual, 2003). These dimensions are illustrated in Figure 3.5. Bankfull height (from bed level to the level of maximum within-bank stage), rather than banktop height, is more commonly used to describe channel shape but this is not recorded in the RHS. Banktop height is recorded separately for both the left (LBTH) and right bank (RBTH) and the two were averaged to obtain a single height measurement more representative of the 500 m reach. In addition, water depth, which is not included in banktop height, was added to the banktop height to give a measure more representative of channel shape. This new composite measure (Equation 3.1) was used in conjunction with bankfull width to obtain the width depth ratio:

$$\text{Width Depth Ratio} = W / (((RBTH + D) + (LBTH + D))/2) \quad (3.1)$$

Where:

- W = Channel Width,
- RBTH = Right Banktop Height
- LBTH = Left Banktop Height
- D = Water Depth

Any survey entry which lacked data in any of the four contributing variables returned an invalid width depth ratio result and these were removed before analysis. As an aid to interpreting the width depth ratio index at a site, note that any values above 1.0

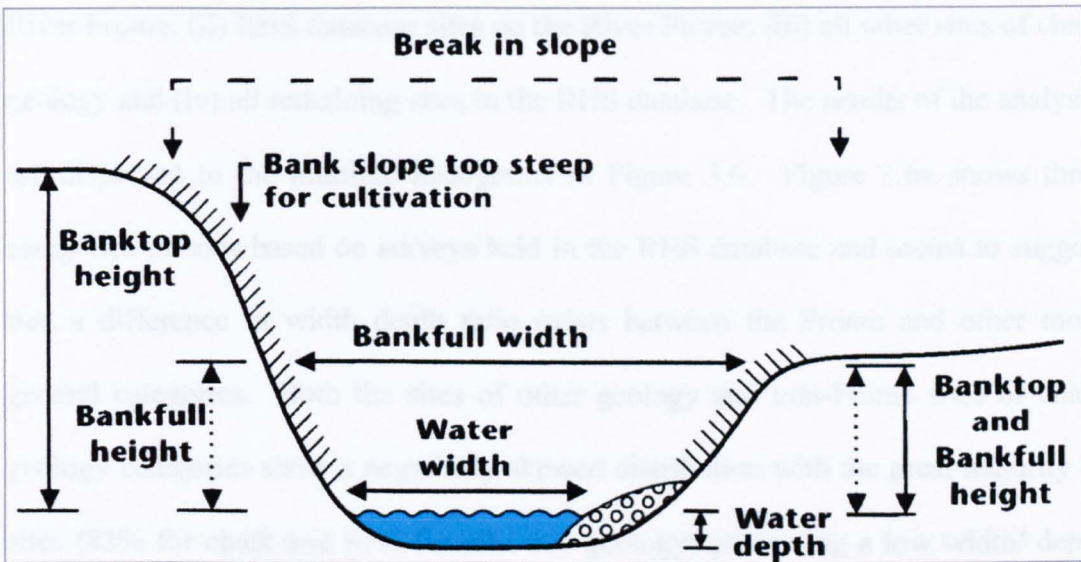


Figure 3.5 An illustration of the RHS channel dimensions, bankfull width and banktop height, used to derive the width depth ratio for sites within the RHS and LOCAR databases. Adapted from Environment Agency (2003).

indicate that width exceeds depth while values below would 1.0 indicate that depth exceeds width.

3.3.1 Comparisons of channel shape

The width depth ratio was calculated for each viable site in the RHS and LOCAR database and sites were divided into four categories: (i) LOCAR database sites on the River Frome; (ii) RHS database sites on the River Frome; (iii) all other sites of chalk geology and (iv) all remaining sites in the RHS database. The results of the analyses are displayed in the multiple histograms in Figure 3.6. Figure 3.6a shows three categories of sites based on surveys held in the RHS database and seems to suggest that a difference in width depth ratio exists between the Frome and other more general categories. Both the sites of other geology and non-Frome sites of chalk geology categories show a negatively skewed distribution with the great majority of sites (83% for chalk and 87% for all other geology) possessing a low width/ depth ratio of less than 10. The Frome by contrast exhibits a more uniform distribution with only 52% of sites with a width depth ratio <10, while 30% of RHS database sites on the Frome have a width depth ratio of 10 - 15. The Frome also has a much smaller range of values than the two larger chalk and other geology types, for example, the Frome has a maximum width depth ratio of 20.7 while the non-chalk sites have a maximum width depth ratio of 116.7. Figure 3.6b provides a comparison of the width depth ratio characteristics of Frome sites in the RHS and LOCAR databases. Both site groupings exhibit a limited range of values and a relatively uniform distribution, although the more extensive LOCAR database has a greater proportion of sites with a width depth ratio under 10 (70%).

3.3.2 Channel shape at the River Frome study reaches

Figure 3.7 illustrates the average width depth ratio for all four site categories. In both the LOCAR and RHS database categories the Frome has a higher width depth ratio than either the other chalk sites or sites of other geology. These latter two have very similar average width depth ratio of 6.76 and 6.09 respectively, while the RHS Frome average scores roughly 45% higher than this at 9.9. This would suggest that, on average, the Frome has a wider, shallower, channel shape than the average for other rivers.

The width depth ratio for representative cross-sections at the study sites were also calculated and are listed in Table 3.3. It is evident that the three study site exhibit a higher width depth ratio than the average value for the River Frome, indicating that the sites are wider and shallower, when compared to their bank heights, than the average for the Frome. However, the study site values do lie well within the range of values obtained for the Frome from the RHS and LOCAR databases.

3.4 Riparian tree cover

Shading by riparian trees restricts light availability in the channel (Pitlo, 1978) and this is thought to be an important control on aquatic vegetation growth: the planting of riparian trees has been suggested by several authors as an alternative form of vegetation management. This section compares the degree of shading in chalk rivers to that of other geologies and to shading on the river Frome. The degree of riparian shading on the Frome as a whole is also compared to shading in the study reaches.

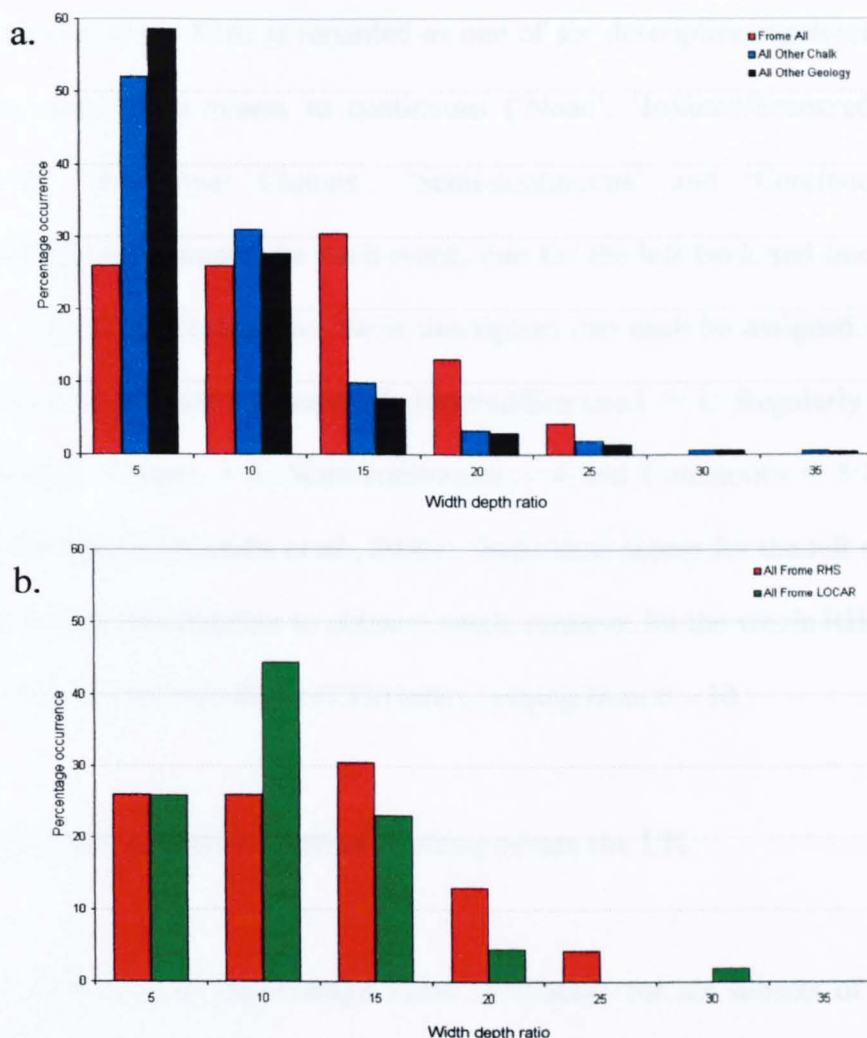


Figure 3.6 Multiple histograms showing the distribution in width depth ratio for: a) three subsets of the RHS database (1994 – 2002); and b) a comparison of River Frome sites from the RHS database and LOCAR database (2003).

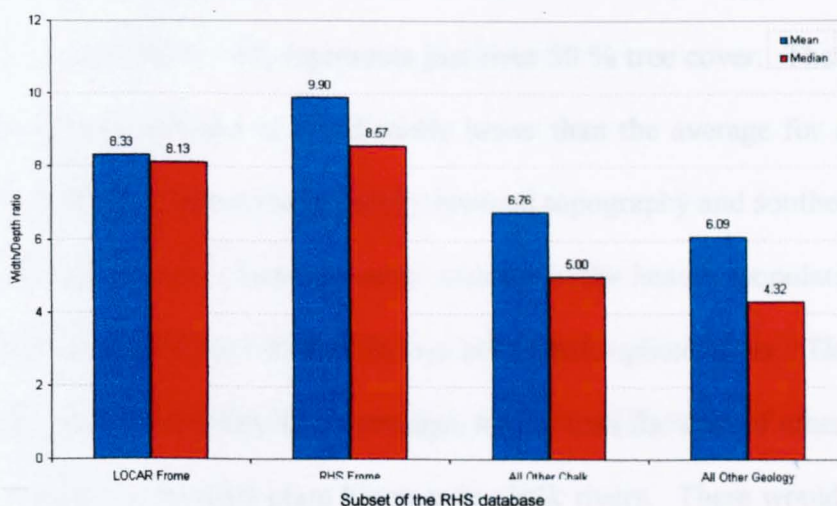


Figure 3.7 Mean and median width depth ratio for three subsets of the RHS database (1994 – 2002) and sites on the River Frome from the LOCAR database (2003)

Tree cover in the RHS is recorded as one of six descriptive reach-scale categories which range from absent to continuous ('None', 'Isolated/Scattered', 'Regularly spaced', 'Occasional Clumps', 'Semi-continuous' and 'Continuous'). Two descriptors are assigned for each reach, one for the left bank and one for the right bank. To facilitate analysis, these descriptors can each be assigned a quantitative value or 'tree score': None = 0; Isolated/Scattered = 1; Regularly Spaced = 2; Occasional Clumps = 3; Semi-continuous = 4 and Continuous = 5 (based on the methodology of Boitsidis *et al.*, 2006). Individual scores for the left and right bank are simply added together to obtain a single measure for the whole RHS reach. This provides a Total Tree Score (TTS) index ranging from 0 – 10.

3.4.1 A comparison of riparian shading across the UK

Figure 3.8 displays the average Total Tree Score for six subsets of the RHS and LOCAR: chalk sites on the River Frome; non-chalk sites on the Frome; all sites on the Frome; all other chalk sites; chalk sites including the River Frome; and sites of all other geology. The average TTS for UK sites of other geology in the UK is 5.5, which, on a scale of 0 – 10, represents just over 50 % tree cover. The average TTS for chalk sites is 4.6 and is significantly lower than the average for sites of other geology. This may reflect the generally lowland topography and southern location of most chalk catchments - less tree cover remains in the heavily populated and urban, southern lowlands of the UK than in less populated, upland areas. This lower TTS means that light availability is, on average, higher than for sites of other geology and may contribute to the high plant biomass in chalk rivers. There would appear to be

potential for the control of aquatic vegetation growth by manipulating riparian tree cover.

Differences in lowland and upland locations may also be observed in the two TTS values for the River Frome. The Frome displays the opposite of the UK picture: TTS is significantly higher for chalk sites on the Frome than for the non-chalk sites. This may be explained with reference to the geological map in Figure 2.2, Chapter 2. Chalk outcrops in the central portion of the Frome catchment where land use is more rural and where tree cover is higher. In contrast, the non-chalk sites are located in the lower reaches of the Frome and are more affected by urban land use, and have a lower TTS score. The average TTS for Frome chalk sites is much higher than the average TTS for all other UK chalk sites. This may again be attributed to the more upstream location of the Frome chalk sites. The overall TTS for the Frome (chalk and non-chalk sites combined) is the same as the average for all other UK chalk sites.

3.4.2 Riparian shading at the River Frome study reaches

The study sites are located in the rural, chalk area of the Frome catchment and may be expected to experience slightly higher shading than the chalk average. Total Tree Scores computed from primary RHS surveys are shown in Table 3.3 and demonstrate significant site differences. The Crockways (TTS = 5) and Frampton site (TTS = 6) are very close to the mean and median total tree scores for chalk sites on the Frome and represent typical sites. The Hydeclift plantation site, however, has a TTS of 8

which is not typical of the Frome. It is likely that m-chained vegetation will be limited at this site compared to other sites on the Frome.

3.3 In-channel vegetation

Understanding the nature of m-chained vegetation in the Frome is fundamental to

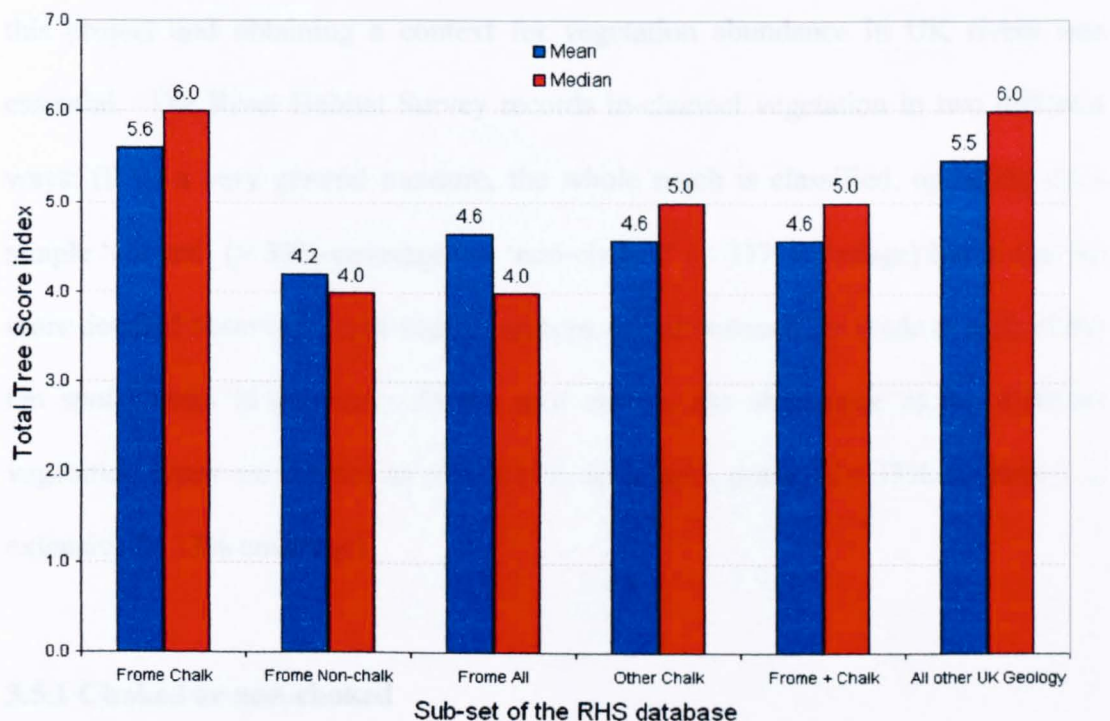


Figure 3.8 Total Tree Score (TTS) Index for six subsets of the RHS and LOCAR databases.

which is not typical of the Frome. It is likely that in-channel vegetation will be limited at this site compared to other sites on the Frome.

3.5 In-channel vegetation

Understanding the nature of in-channel vegetation in the Frome is fundamental to this project and obtaining a context for vegetation abundance in UK rivers was essential. The River Habitat Survey records in-channel vegetation in two different ways: (i) as a very general measure, the whole reach is classified, operating on a simple ‘choked’ (> 33% coverage) or ‘non-choked’ (< 33% coverage) basis; and (ii) more detailed observations of vegetation type and abundance are made at each of the ten spot checks in a reach. At the spot checks, the abundance of ten different vegetation types are classed as absent (0% coverage), present (< 33% coverage) or extensive (> 33% coverage).

3.5.1 Choked or non-choked

Figure 3.9 shows a PCA plot showing all rivers in the current database classified as choked and non-choked (denoting vegetation abundance above and below 33% coverage). The non-choked river sites are larger in number and retain the approximate shape and distribution of the main, overall data body while the choked sites have a more restricted location and distribution. The location of the choked sites is similar to that of the chalk sites in Figure 3.2 and most sites are located in the lowland, coastal and low-energy upland sites. However, there is quite a lot of scatter in the choked data and several sites occur in a montane setting. An analysis of the

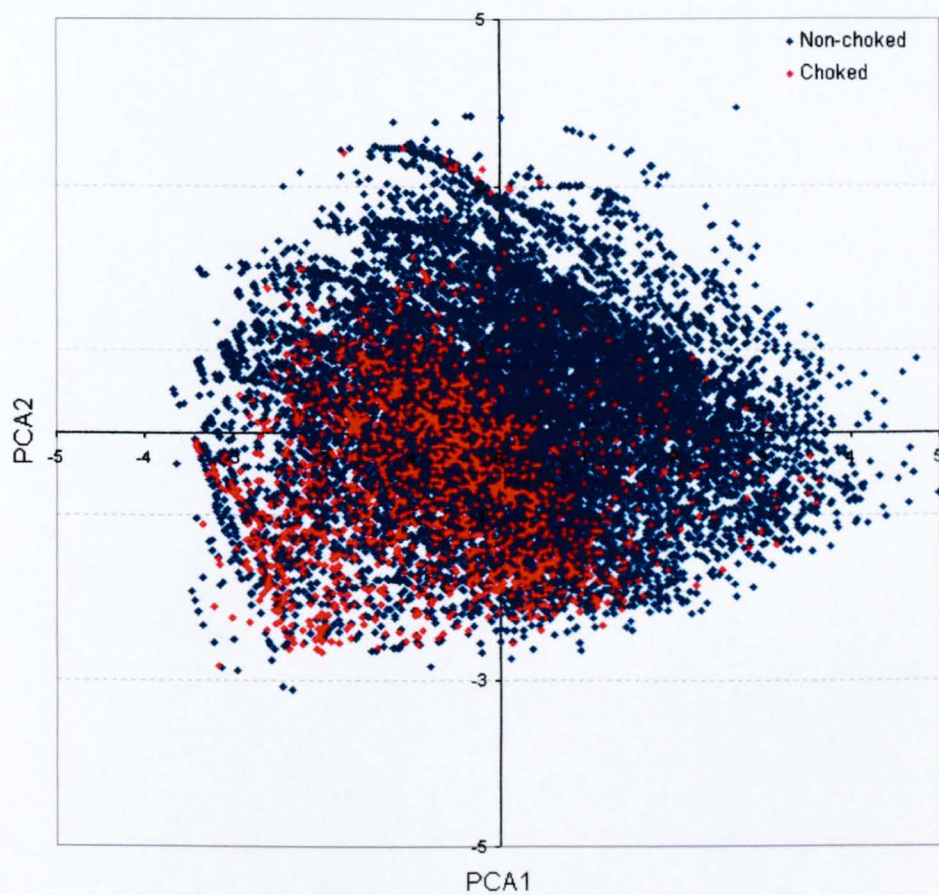


Figure 3.9 PCA plot showing the plot location of choked and non-choked sites from the RHS database (1994 – 2002).

‘choked’ and ‘non choked’ data reveals that only 10.6 % of the total database sites are classed as choked; in non-chalk sites this is reduced to 9.4 % while in chalk sites 23.3 % are choked. This indicates that chalk river sites have, on average, a greater proportion of ‘choked’ high biomass sites than the average for sites of other geology.

3.5.2 Spot-check based in-channel vegetation: calculation of the in-channel vegetation index (ICV)

The vegetation at RHS spot checks is recorded using ten categories broadly based on either plant morphology, e.g. ‘Submerged broad-leaved’ and ‘Submerged linear-leaved’, or on more species-based criteria e.g. ‘Liverworts/mosses/lichens’ and ‘Filamentous algae’. Each vegetation type is given an abundance rating at each spot check: either absent (0% coverage); present (<33% coverage) or extensive (> 33% coverage). These separate vegetation groupings and abundance ratings are very informative at a site-specific scale of analysis, but they must be simplified before use in more general analyses of the whole database. The methodology of Emery (2003 and 2004) was used to reduce all spot-check vegetation information to a single representative index of vegetation abundance for each site. In this method, each vegetation type is assigned to one of three broader categories, formed according to expected plant biomass and relative influence on flow velocity. Specific weightings are given to each category and each level of abundance.

Three vegetation types: ‘Liverworts/mosses/lichens’; ‘free floating’ and ‘amphibious’ (trailing riparian) vegetation, were deemed by Emery to have limited biomass, and negligible impact on flow, and were excluded from the overall index

(Emery, 2003). The other seven vegetation categories were divided into groups of ‘intermediate’ and ‘major’ influence. The weightings for each vegetation group and abundance rating are given in Table 3.4. The RHS can also record vegetation at a site as ‘not-visible’ at very deep sites or in turbid conditions; any observations reported as not visible were removed from the data set before analysis. The three plant groups with negligible influence are all given weightings of zero, which excludes them from calculations of the overall vegetation index. Seven separate equations were required to combine the weightings of the remaining vegetation categories into a single in-channel vegetation index. These seven equations are given below and summarised in Equation 3.2:

Floating leaved (FL)	=	$(0.5 \times P + 1.0 \times E) / (A + P + E)$
Submerged linear-leaved (SL)	=	$(0.5 \times P + 1.0 \times E) / (A + P + E)$
Filamentous algae (FA)	=	$(0.5 \times P + 1.0 \times E) / (A + P + E)$
Emergent broad-leaved (EB)	=	$(1.5 \times P + 3.0 \times E) / (A + P + E)$
Emergent reeds/sedges/... (ER)	=	$(1.5 \times P + 3.0 \times E) / (A + P + E)$
Submerged broad-leaved (SB)	=	$(1.5 \times P + 3.0 \times E) / (A + P + E)$
Submerged fine-leaved (SF)	=	$(1.5 \times P + 3.0 \times E) / (A + P + E)$

In-channel vegetation index (ICV)	=	$FL + SL + FA + EB + ER + SB + SF$
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(After Emery, 2003 and 2004)	(3.2)
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A (absent), P (present) and E (extensive) correspond to the number of spot checks in a reach at which each vegetation type was recorded.

Table 3.4 Weightings for RHS in-channel vegetation types used in the calculation of a single in-channel vegetation index (ICV). Adapted from Emery (2003).

Grouping	Vegetation Type	Weighting		
		Absent	Present	Extensive
Negligible	Liverworts/mosses/lichens	0	0	0
	Free-floating	0	0	0
	Amphibious	0	0	0
Intermediate	Floating leaved (rooted)	0	0.5	1.0
	Submerged linear-leaved	0	0.5	1.0
	Filamentous algae	0	0.5	1.0
Major	Emergent broad-leaved herbs	0	1.5	3.0
	Emergent reeds/sedges/ rushes/grasses/horsetails	0	1.5	3.0
	Sunmerged broad-leaved	0	1.5	3.0
	Submerged fine-leaved	0	1.5	3.0

3.5.3 ICV comparisons

The in-channel vegetation index was calculated for each site in the RHS database and for the Frome LOCAR sites. Figure 3.10 illustrates the mean and median ICV for nine subsets of the RHS and LOCAR databases. The most immediate and simple observation is that the first five categories on the left have significantly higher average ICV indices than the four on the right of the graph. These first five series represent river sites either underlain by chalk geology, or sites influenced by chalk elsewhere in their catchment, indicating that chalk rivers have a higher average ICV score than the average for other rock types. This geological division is best emphasised using the mean values for ‘all chalk sites’ compared to the mean ICV for ‘all sites of other geology’; the chalk sites have an average ICV of 3.1, 183% higher than the average for the other geology sites at 1.7. The average for the River Frome falls marginally below the average for all chalk sites at 2.8 and one would assume this was due to the non-chalk sites included in this average or it could be related to a higher level of riparian shading on the Frome, as evident in Section 3.5. However, comparisons of chalk and non-chalk sites on the River Frome show little differentiation between the two geology types. This indicates that the chalk dominance of the catchment may have a significant influence upon the vegetation at individual river sites dominated by another rock type: though not underlain by chalk rock these sites exhibit a ‘chalk’ vegetation signal. As a side issue, the ICV for sites of other geology in Scotland and Northern Ireland is well below the average for England and Wales, perhaps as a consequence of generally cooler climates.

The RHS sites were arbitrarily divided on the basis of their ICV scores and plotted on the PCA plot (Figure 3.11). The number of observations in each category

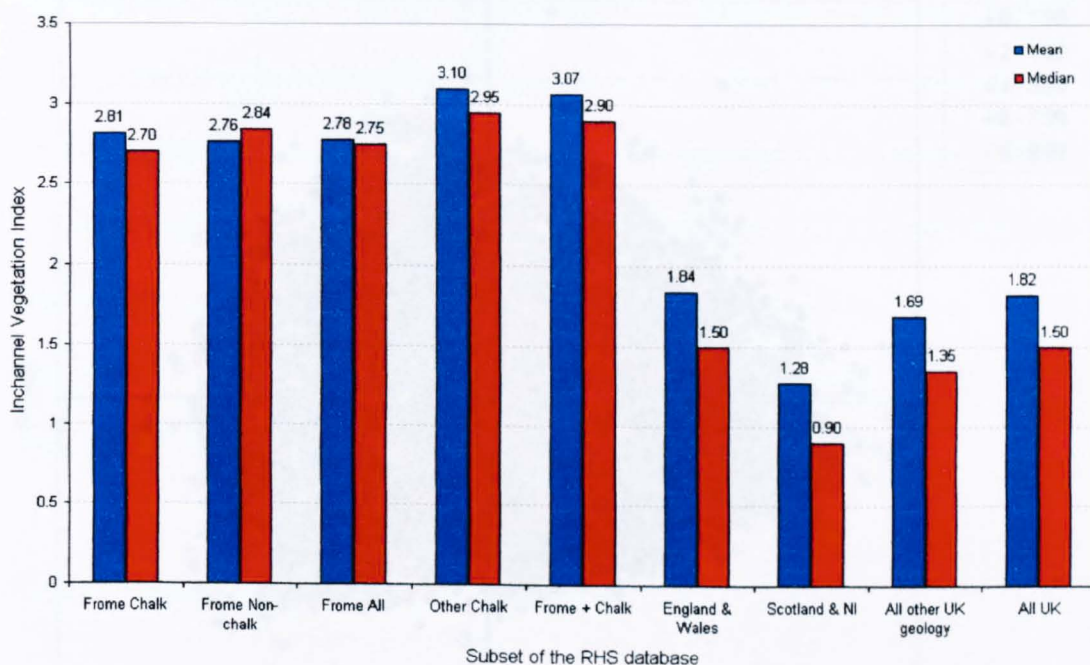


Figure 3.10 A comparison of mean and median in-channel vegetation index (ICV) for nine subsets of the RHS database (1994 – 2002).

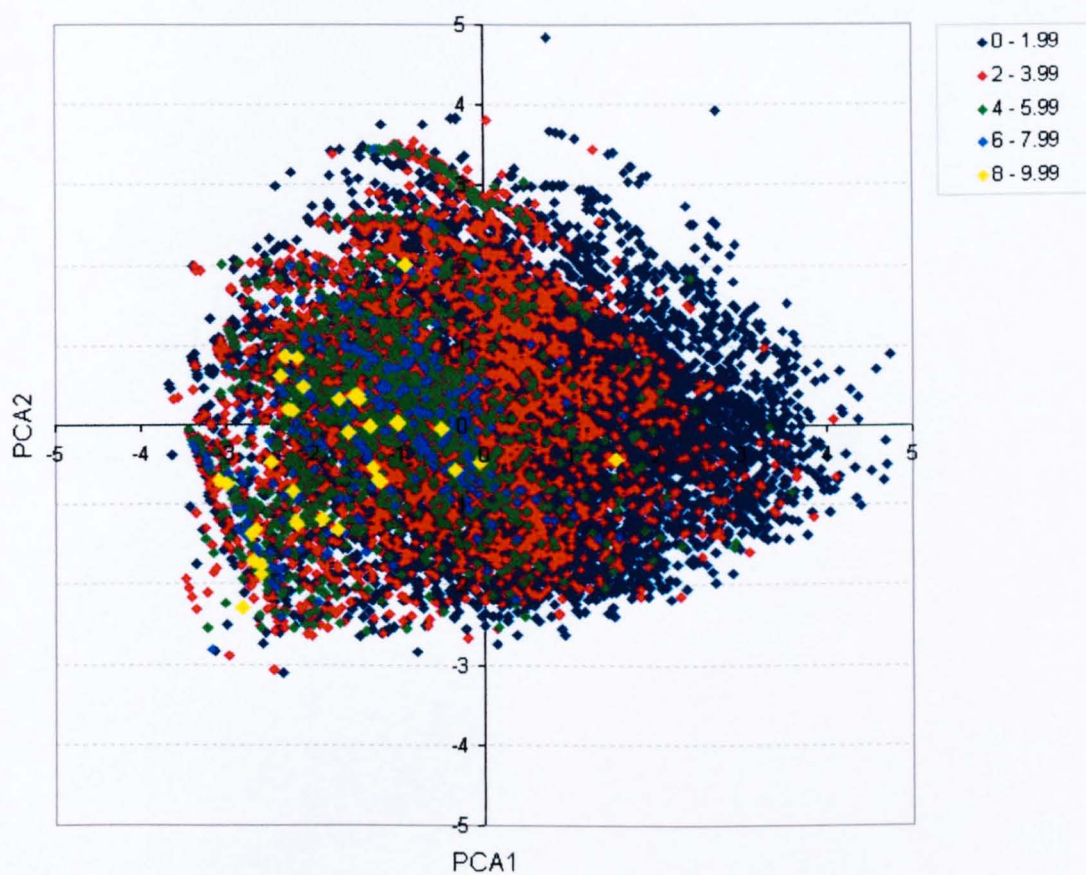


Figure 3.11 PCA plot of all sites in the RHS database (1994 – 2002) subdivided according to in-channel vegetation (ICV) index.

declined with increasing ICV and comparisons should be treated with caution, but, nevertheless, the plot showed some interesting differences. On the whole, the centre of the datasets plotted progressively further to the left of the PCA map with increasing ICV, indicating that changes in PCA1 (altitude and slope) may influence vegetation abundance. It would appear that sites of lower altitude and lower slope have higher ICV scores. This is not unexpected, as vegetation is generally thought to be more abundant in lowland than in upland reaches. Also, the spread of values on the y-axis (PCA2) is generally more constricted for the higher ICV scores. This may be a consequence of decreasing sample size but could also indicate that incidents of high vegetation abundance are concentrated at intermediate-energy sites. This is in agreement with the habitat preferences of many aquatic vegetation species (see Appendix 1).

3.5.4 In-channel vegetation at the River Frome study sites

The in-channel vegetation indices for the three study sites on the Frome, in four seasons, are displayed in Figure 3.12. The most immediate observation is that the Frampton Estate site has a much higher ICV index than either Crockways or Hydeclift in all seasons. The ICV at Frampton in summer (4.85) and in autumn (4.4) is also significantly higher than the summer ICV for the UK chalk average (3.07) and for the average for the River Frome (2.78) and represents a site of very high vegetation biomass. Any vegetation influence on velocity and stage observed at this site may be extreme compared to the vegetation effects at other sites on the Frome and at other chalk sites and this must be considered if results from this study are applied to other sites or catchments. The two upstream study sites, Crockways and

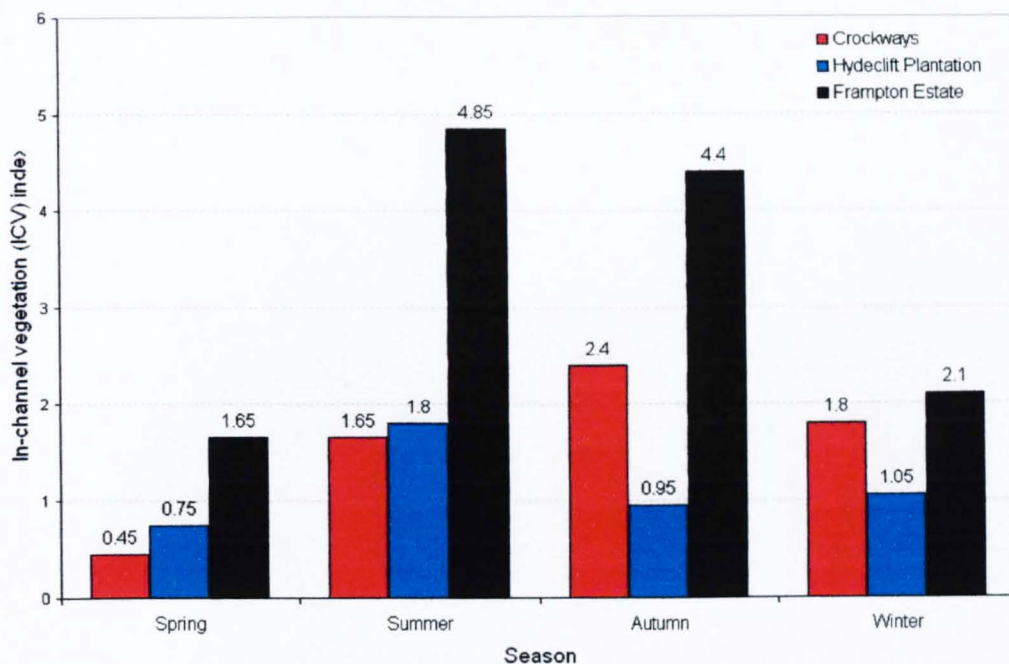


Figure 3.12 Seasonal In-Channel Vegetation (ICV) scores for the three study sites on the River Frome, based on data from primary RHS surveys carried out in April, July, and October 2004 and January 2005.

Table 3.5 Categories used in the RHS survey to record the predominant bed substrate at spot checks and the Mid-point phi rating (based on Wentworth, 1922) assigned to each category in order to calculate the substrate index for each 500m RHS reach.

RHS substrate type	Mid-point phi rating
Not Visible (NV)	N/A
Bedrock (BE)	N/A
Boulder (BO)	- 10
Cobble (CO)	- 7
Pebble (P)	-5
Gravel/Pebble (GP)	- 2
Gravel (P)	- 1.5
Sand (SA)	+ 2
Silt (SI)	+ 5
Clay (CL)	+ 10
Peat (PE)	N/A
Earth (EA)	N/A
Artificial (AR)	N/A

Hydeclift, display a much more moderate ICV index, with values well below the UK chalk average, and below the average for the River Frome, in both summer (1.65 and 1.8) and autumn (2.4 and 0.95). These sites represent moderate to low biomass sites (relative to other chalk sites) and any observed effects on stage, velocity or sediment should be considered in this context. It is interesting to note that the Hydeclift Plantation site has a higher summer ICV score than the Crockways site, despite the marked differences in total tree score observed in Section 5.3.3, and this suggests that riparian shading may not be the only major control on vegetation growth.

3.6 Bed Substrate

In a similar manner to the in-channel vegetation, channel substrate is recorded in the RHS survey as categorised observations at individual spot checks. Thirteen categories are used in the RHS and these are detailed in Table 3.5. Only one predominant category is recorded at each spot check but this still yields ten observations for each RHS site, i.e. one at each spot check. These ten observations must be combined to create one representative index before attempting any quantitative analysis of such a large number of sites and a modification of Emery's method (2003) was used. Emery argued that many of the categories used as RHS substrate descriptors are consistent with the substrate types of the Wentworth scale, a commonly used classification of substrate type, and this idea is replicated here. The Wentworth classification was utilised to obtain physically representative weightings for each RHS substrate category, but the chosen weightings differ from those used by Emery (2003). The Wentworth scale gives a range of phi values for each substrate class and it was decided best to use the mid-point phi rating for each category. These

weightings are detailed in Table 3.5. Of the 13 original categories five are not suitable for inclusion in the index and were excluded before calculations took place; these include the category ‘not visible’ and four others that represent a surface (e.g. bedrock) rather than a substrate (Table 3.5). The Substrate Index for each RHS site is calculated according to Equation 3.3 below:

Substrate index =

$$\frac{(-10 \times \text{BO} + -7 \times \text{CO} + -5 \times \text{P} + -2 \times \text{GP} + -1.5 \text{G} + 2 \times \text{SA} + 5 \times \text{SI} + 10 \times \text{CL})}{(\text{BO} + \text{CO} + \text{GP} + \text{SA} + \text{SI} + \text{CL})} \quad (3.3)$$

(Modified from Emery, 2003).

Where BO (boulders), CO (cobbles), GP (gravel/pebble), SA (sand), SI (silt) and CL (clay) are equal to the number of spot checks at which each sediment calibre was recorded.

The substrate index represents an average measure of several sediment types and cannot be directly compared to a phi size unless all spot checks at a site have recorded the same substrate type. However, the index can be seen as an indication of average sediment grade, and forms a continuum between 10 and –10 (representing exclusively clay and exclusively boulders respectively); sediment scores may be judged relative to each other. Multiple histograms displaying the distribution of the sediment index are displayed in Figure 3.13 for: (a) three subsets of the RHS database (sites on the Frome; sites of chalk geology and all sites of other geology);

and (b) Frome sites from the RHS and LOCAR database. Figure 3.13a reveals that chalk sites have a generally coarser sediment grade than the sites of other geology: chalk sites are not represented in either of the smallest sediment size classes (-10 to -8 phi and -8 to -6 phi) and the chalk distribution plots generally more to the right than does the sites of other geology, indicating coarser substrate. Figure 3.13a also indicates that the River Frome has a narrow distribution in terms of the sediment index (sediment index -6 to 4). This narrow distribution for RHS Frome sites is mirrored by the LOCAR sites on the Frome in Figure 3.13b. Figure 3.14 shows the substrate index for the River Frome displayed with reference to longitudinal position along the river. This reveals that the substrate in the Frome is generally of coarse grade, with all but one value falling below zero. The substrate grade is also fairly consistent along the length of the River Frome; sites with coarser substrate grades exist but in general the index rarely falls below -2.

3.6.1 Bed substrate at the River Frome study reaches

Table 3.3 details the substrate index values for the three study sites in July 2004. The summer survey was chosen because it afforded the best view of substrate at low flow and low turbidity. All three sites have a negative substrate index and this is in keeping with the coarse substrate of the Frome RHS and LOCAR sites. However, there are significant differences between the three sites. The Hydeclift site has a substrate index of -5.1, which is coarser than any of the sites in the LOCAR or Environment Agency RHS sites on the Frome, and this suggests that this is a high energy site of low morphological and substrate diversity which is not common on the Frome. The Crockways (-1.1) and Frampton reaches (-2.8) have substrate index

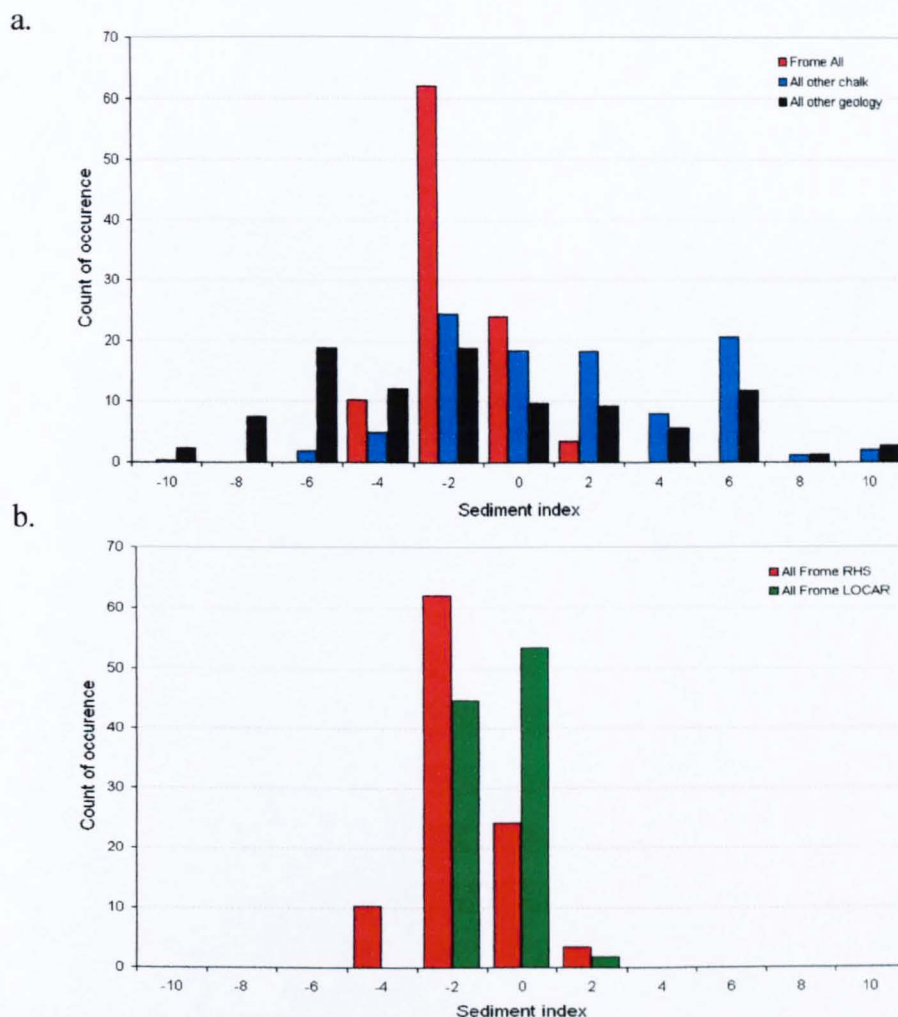


Figure 3.13 Multiple histograms showing the distribution of the sediment index for: a) three subsets of the RHS database: sites on the Frome; chalk sites and all sites of other geology; and b) for the Frome from the RHS and LOCAR databases.

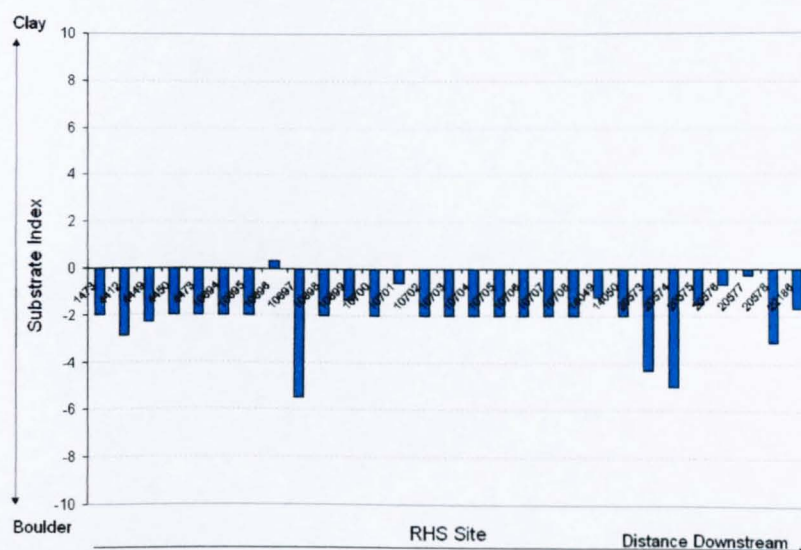


Figure 3.14 Bar chart showing the substrate index at sites on the Frome from the RHS database. This demonstrates the coarse nature of bed sediments in the Frome from source to mouth.

values much more in keeping with the typical values of the Frome and this demonstrates the greater morphological diversity of these reaches which contain several pools and riffles. The Crockways site is the most morphologically diverse, and has the highest substrate index score.

3.7 Discussion: the chalk sub-group

The analysis presented in this chapter provides a quantitative exploration of the complete geographical and climatic range of English chalk rivers and confirms the established concept of chalk rivers as a distinct UK-river sub-group, with distinguishing in-channel features. Rivers in the chalk group may be defined through the River Habitat Survey analysis as having a lower source height, slope and altitude than rivers of other geology and consequently a lower energy regime. Chalk rivers may also be defined as having generally lower levels of riparian shading, a slightly higher width depth ratio and coarser substrate than rivers of other geology. These factors, along with abundant nutrient supply from the chalk rock and, generally southern, warmer climates mean that chalk rivers have a demonstrably higher vegetation biomass than other rivers. The RHS analysis has shown that the average in-channel vegetation index of 3.1 for chalk river sites was demonstrably higher than the average for non-chalk river sites of 1.69, a difference of 183%. Very high biomass sites, or sites that might be considered ‘problematic’, were also of higher incidence in chalk rivers: 23.3 % of chalk rivers were classed as ‘choked’ (> 33 % vegetation cover) compared to only 9.4 % for rivers of other geology. Together with the data and descriptions obtained from the literature in Table 3.1, the chalk river sub-group may be defined under several criteria, as set out in Table 3.6. Chalk rivers

Table 3.6 Summary descriptors of the chalk river group and the River Frome with reference to previous research and to new analysis of the national RHS database. An asterisk (*) denotes descriptors which have been strengthened by input from the RHS database analysis.

Descriptor	The chalk group	The River Frome
Discharge regime	Stable river flows; major flooding rare; predictable seasonal discharge pattern: min flow in summer, max in winter. Max to min flow of 10:1.	Stable river flows; major flooding rare; predictable seasonal discharge pattern: min flows in summer, max in winter.
Energy regime*	Lower average altitude, slope and source height than average for sites of other geology and consequently a lower energy regime.	Higher altitude, slope and source height than the chalk group average and consequently a higher energy regime.
Fauna	Diverse and productive communities of high biomass; each plant species supports a specific faunal assemblage.	Diverse and productive communities of high biomass; each plant species supports a specific faunal assemblage.
Flora*	Summer plant biomass may reach 400 g dry weight m ⁻² , but typically 200g dry weight m ⁻² ; longitudinal change in plant species type from source to mouth. Average ICV index much higher for chalk group than for sites of other geology (3.1 compared to 1.69) and much higher percentage of high biomass 'choked' sites (23.3 % compared to 9.4 %).	Average biomass 200g dry weight m ⁻² ; ranging from 30g dry weight m ⁻² in deep water to 300 – 400 g dry weight m ⁻² in shallow water; longitudinal change in plant species type from source to mouth. Lower average ICV index than the chalk group average (2.78 compared to 3.1).
Human impacts	Clearance of riparian tree cover; water meadows and supply channels; commercial watercress farms and fishing; vegetation cutting and removal for flood prevention; groundwater and surface water abstraction.	Clearance of riparian tree cover; water meadows and supply channels; commercial watercress farms and fishing; vegetation cutting and removal for flood prevention; groundwater and surface water abstraction.
Riparian shading*	Riparian shading at chalk sites lower than the average for sites of other geology. May reflect location of chalk rivers in UK: less tree cover remaining in the heavily populated and urban, southern lowlands of the UK.	Riparian shading significantly higher for chalk sites on the Frome than for non-chalk sites. Chalk outcrops in upper portion of catchment where land use is rural and tree cover is higher; non-chalk sites found in lower reaches where land use is more urban.
Substrate and sediment*	Coarse gravel and cobble beds with flints and concretion; discrete deposits of fine, organic sediment beneath plant beds.	Coarse gravel and cobble beds with flints and concretion; discrete deposits of fine, organic sediment beneath plant beds.
Water quality	Low turbidity; high nutrient availability; pH 7.4 – 8.5; maximum seasonal temperature range of 2 – 25°C and monthly mean variation < 8°C; 83 % of chalk rivers of good chemical quality (2000).	High levels of phosphate and nitrate; pH 7.4 – 8.4; temperature ranges from 5.5 – 7.5°C in January to 16.0 – 18.5°C in August. Range of annual means (1969 – 1972) of 10.4 – 10.8°C.
Width depth ratio*	Average and distribution of width depth ratio similar to sites of other geology (average 6.76 compared to 6.09).	Width depth ratio significantly higher than the chalk group average (9.9). Distribution more even, indicating stability in cross-section shape along the river.

share many of the same physical and ecological characteristics and this will likely impose similar management challenges and will require similar solutions.

3.7.1 The position of the River Frome within the chalk sub-group

While confirming the general consensus and picture of the chalk rivers, the RHS analysis also suggests that the Frome is not a typical or average example within the chalk group. The Frome has a higher slope and source height than many of the chalk rivers and consequently has a higher energy regime and relatively less vegetation growth than the average for other chalk rivers. The average in-channel vegetation index for the River Frome is 2.78 compared to an average of 3.1 for the chalk group as a whole. The Frome average is compared with the chalk average for a number of parameters in Table 3.6 and differs under several of these parameters. However, the Frome is still demonstrably a member of the chalk group and though it is not, in some respects, representative of the chalk average, it is sufficiently close to be considered as a member of the chalk group, especially in terms of high in-channel vegetation index. Research findings derived from work on the River Frome should be applicable to other chalk rivers, but with the understanding that some characteristics are different to the chalk average, in particular energy regime and channel shape.

3.7.2 Differences between the study sites and comparisons to the Frome average

The chosen study sites on the Frome were also examined using primary RHS surveys, collected as part of the field campaign for this thesis, in order to examine the

typicality of the sites when compared to the Frome average and when compared to each other. Overall, the study sites are generally close to the Frome average in terms of energy regime but provide interesting contrasts to the River Frome average and to each other in terms of width depth ratio, total tree score, in-channel vegetation and bed substrate.

The total tree score and in-channel vegetation index are particularly important in the context of this thesis. The total tree score differed markedly between sites: the Crockways and Frampton sites were very close to the Frome average, while the Hydeclift Plantation site exhibited very strong riparian shading compared to the Frome average, which may greatly inhibit vegetation growth at the site. The in-channel vegetation index also differs markedly between sites, but not in the manner expected from the total tree score values, for example the summer ICV index at the moderately shaded Crockways site was less than at the heavily shaded site at Hydeclift. This suggests that riparian shading is not the only major control on vegetation cover in a reach. The summer ICV index values for the Crockways and Hydeclift sites were below average for the river Frome, while the ICV index for the Frampton site was well above average. These marked differences between study sites should provide interesting contrasts in vegetation influence and should provide information as to the physical controls that determine vegetation abundance and influence.

4. SEASONAL TRENDS IN REACH SCALE HYDRAULICS

4.1 Chapter Synopsis

This chapter aims to determine whether cycles of aquatic vegetation growth and decline have an appreciable effect on hydraulic roughness, water surface elevation and water surface slope at sites on the River Frome. Gross seasonal changes in the stage/discharge relationship and in hydraulic geometry relationships are examined, and an attempt is made to separate the two linked processes which contribute towards seasonal change in river stage: (i) the reduction of water velocity within plant beds due to increased hydraulic resistance; and (ii) seasonal changes in sedimentation, or 'scour and fill', induced by the vegetation. Seasonal changes in river stage are quantified for two contrasting sites on the River Frome (the Crockways and Frampton study sites), and estimates of the relative importance of the two contributing factors are given for each site. Comparisons are also made as to the changing influence of vegetation on stage at high and low discharges. The effects of vegetation growth and decay on water surface slope are also investigated, and are considered at both long-term, seasonal, and shorter, event-based, time-scales. The impact of vegetation growth on stage is contrasted between individual sampling locations at the same site, and between two sites of very different morphology, to illustrate the effect of vegetation in modifying the signals imposed by local planform and in-channel features. Throughout the chapter consideration is given to the implications of the research findings for current and alternative management practices.

4.2 The influence of macrophytic vegetation on roughness, velocity and sedimentation, and cumulative effects on river stage

Table 1.1, in Chapter 1, summarises the opinions of authors on the nature and magnitude of velocity reductions by aquatic plants in river environments. There is a consensus that large aquatic plants do cause velocity reductions within their beds, but some authors also cite velocity increases outside of plant beds (Sand-Jensen and Mebus, 1996; Gurnell *et al.*, 2006; Cotton *et al.* 2006; Wharton *et al.* 2006). There is divided opinion as to the effect of macrophytes on river stage: some authors believe that the increased roughness and velocity reduction within vegetation beds is sufficient to increase stage levels and the incidence of overbank flooding (Haslam, 1978; Hearne and Armitage, 1993; Dodds and Biggs, 2002), while others consider that velocity increases outside plant beds compensate for reductions within. On either side of the debate, quantitative results are limited, particularly at the meso- and macroscale. This section examines the available literature in more detail and considers the effects of macrophytes on hydraulic roughness, water velocity, sedimentation and river stage.

4.2.1 Hydraulic roughness

Hydraulic roughness refers to the degree of irregularity in a stream channel, the energy loss in overcoming this irregularity, and the effect on mean velocity. Irregularities include: bed substrate; channel sinuosity; in-channel morphology; artificial obstructions; and vegetation. Roughness is measured through a combination of three main factors: velocity, hydraulic radius and water surface slope, and is

generally expressed using a ‘roughness coefficient’. There are a number of coefficients to choose from (Dingman, 1984) but two of the most commonly used are the Manning’s ‘n’ roughness coefficient and the Darcy-Weisbach friction factor, ‘ff’, both of which can be used to approximate bulk changes in reach-scale roughness. The equations used to obtain Manning’s n and Darcy-Weisbach roughness coefficients are given in Equations 4.1 and 4.2 below:

Manning’s roughness co-efficient (n):

$$n = \frac{kR^{2/3}s^{1/2}}{v} \quad (4.1)$$

Darcy-Weisbach friction factor (ff):

$$ff = \frac{8gRs}{v^2} \quad (4.2)$$

Where: k = Constant, either: 1 (SI units) or 1.49 (Imperial units)

v = Mean velocity

R = Hydraulic radius

s = Slope of the energy gradient

g = Gravity constant

Knighton (1998).

Manning’s n and Darcy-Weisbach were chosen as two measures that reflect the total resistance of a reach, including vegetative roughness. These bulk flow roughness parameters are “essentially equivalent and interchangeable” (Dingman, 1984, p. 141) where flow conditions are approximately uniform (i.e. when bed slope and water

surface slope are parallel), but when water slope and bed slope are different and flow is not uniform they are not the same (see Figure 4.1). In addition, Manning's equation is dimensional whereas Darcy-Weisbach is dimensionless. Darcy-Weisbach is often preferred for its dimensionless properties but Manning's n is viewed as the most useful coefficient for estimating the characteristics of flows in natural channels (Dingman, 1984). The coefficients are used in this chapter, as a cross-reference for each other, to illustrate seasonal and site differences.

Macrophytic vegetation adds a dynamic seasonal element to hydraulic roughness. In summer, vegetation growth increases roughness by creating a more varied cross-section and by raising the roughness boundary from the channel bed to the height of the vegetation canopy. Channel roughness declines again to baseline level in winter following the senescence and washout of the plants. For example, Dawson (1978) observed that the Manning's roughness coefficient on the River Piddle, in Dorset, ranged from 0.05 in winter to 0.3 in summer; this equates to a summer increase of 600 % from the winter baseline. Roughness and velocity are intricately linked in all rivers (Dingman, 1984), and it is anticipated that the increase in resistance imposed by macrophyte growth will greatly influence mean cross-section velocity. The analysis in Section 4.5 attempts to confirm this by evaluating the Manning's and Darcy-Weisbach roughness coefficients with respect to river stage, discharge and velocity. However, the roughness coefficients are at best a partial guide to the true effect of vegetation in river channels, as they summarise the influence of many linked contributing factors and their use is not appropriate under all flow conditions.

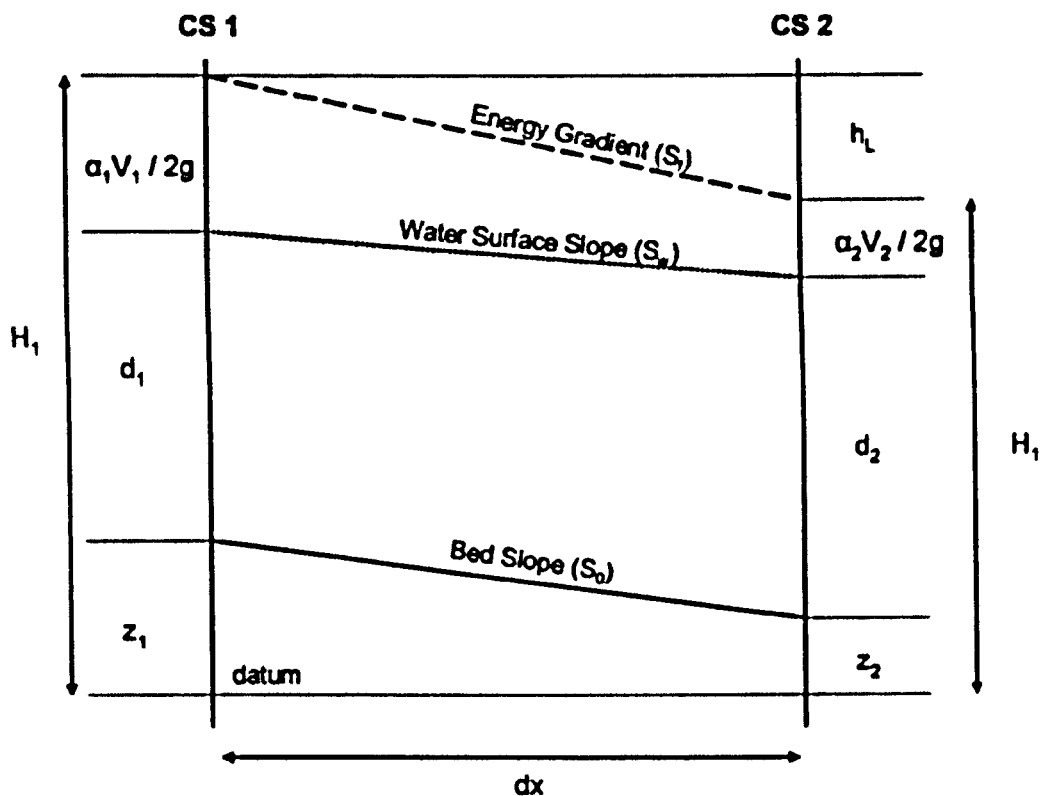


Figure 4.1 Definition diagram for gradually varied flow conditions: depth and velocity vary over long lengths of channel and the energy, water surface and bed slopes are not parallel (after Richards, 1982).

The roughness coefficients are viable under some flow conditions but not under others, and this introduces an element of uncertainty into the analyses. There are three recognised types of water flow in a channel: uniform; gradually varied; and rapidly varied flow (Richards, 1982). Under uniform flow conditions water depth, river discharge, and water velocity are constant with distance along the channel and the bed slope, energy slope and water surface slope are all equal (Figure 4.1). Under varied, or non-uniform, flow conditions depth and velocity change with distance along the channel and the channel slope, the energy slope and the water surface slope are not equal (Figure 4.1). If depth and velocity changes take place over long lengths of channel, e.g. 500 m, then the non-uniform flow may be termed gradually varied, whereas, if abrupt changes in depth and velocity occur over short distances, then flow is termed rapidly varied. The roughness coefficients were designed for use in uniform flow conditions but may be used over short river reaches during gradually varied flow. The uniform flow coefficients may not be used to describe rapidly varied flow (Richards, 1982).

It is not certain that flow conditions recorded in the study reach were always within the range for which the coefficients are applicable, especially at high vegetation cover in summer, and this introduces uncertainty into the analysis. In addition, the roughness coefficients are effectively ‘bulk-flow’ parameters, which take many contributing factors into account, and this means that some elements of the roughness behaviour cannot be resolved and interpretation as to the contributing factors cannot be made. Roughness, velocity, slope, width and depth are “so intimately interrelated that it is impossible to separate cause from effect” (Dingman, 1984 p.137). The n and f coefficients assist in providing an overall appreciation of the seasonal changes

in reach-scale roughness, but the results are rarely fully interpretable. Possible uncertainties in the use of roughness coefficients are minimised by examining the component parts of the roughness equations (velocity, width and depth), which are not invalidated by variations in flow.

In practice, roughness coefficients are often estimated from tabulated values and photographs of representative reaches of known roughness (Richards, 1982). Cowan (1956) presented a procedure for estimating n in natural rivers that clearly demonstrates the many factors that combine to determine flow resistance in a reach. This procedure is summarised below in Equation 4.3:

$$n = (n_0 + n_1 + n_2 + n_3 + n_4) m \quad (4.3)$$

A description of each n component, and suitable values for each component, are listed in Table 4.1 and Cowan's equation is used later in Section 4.5 to help place results from the present study in context.

4.2.2 Velocity

Velocity forms an integral component of the Manning's and Darcy-Weisbach roughness coefficients (Equations 4.1 and 4.2) and, as macrophytic growth is thought to increase hydraulic roughness, most authors agree that velocity will be significantly reduced within macrophytic beds (see Table 1.1, Chapter 1).

Vegetation induced changes in mean channel velocity are predicted to have an impact on river stage and this stems from the continuity equation:

$$Q = wdv \quad (4.4)$$

Where: Q = discharge
 w = water width
 d = water depth
 v = velocity

At a constant discharge a change in mean velocity will necessarily require a change in cross-sectional area: predominantly occurring as a change in water depth, and, generally less markedly, a change in water width. For a given discharge, and with a vegetation-induced reduction in velocity, it would seem likely that a higher water surface elevation will be recorded in summer than for the same discharge in winter. River stage will only be affected by velocity reductions if *mean* reach velocity is altered and there is some uncertainty as to whether vegetation effects are sufficient to cause this. Water velocities within plant beds are known to be reduced, but, equally, it is thought that water velocities are significantly increased in un-vegetated space above and alongside plant stands. Sand-Jensen and Mebus (1996) quantified this process in a Danish stream, and estimated that, at a maximum plant cover of the submerged macrophyte *Callitriche cophocarpa*, only 21% of river discharge passed through the vegetation, while 79% was transported through 'flow channels' between vegetation stands at 2.6-fold higher flow velocity. Velocity increases *above* plant stands have also been investigated by Sand-Jensen and Mebus (1996); Sand-Jensen

Table 4.1 Values for calculating Manning’s n in natural streams using Cowan’s method (1956) (after Chow, 1959).

<i>Channel conditions</i>	<i>Value of n</i>
<i>Material involved</i>	<i>n₀</i>
Earth	0.020
Rock cut	0.025
Fine gravel	0.024
Coarse gravel	0.028
<i>Degree of irregularity</i>	<i>n₁</i>
Smooth	0.000
Minor	0.005
Moderate	0.010
Severe	0.020
<i>Variations of channel cross-section</i>	<i>n₂</i>
Gradual	0.000
Alternating occasionally	0.005
Alternating frequently	0.010 - 0.015
<i>Relative effect of obstructions</i>	<i>n₃</i>
Negligible	0.000
Minor	0.010 – 0.015
Appreciable	0.020 – 0.030
Severe	0.040 – 0.060
<i>Vegetation</i>	<i>n₄</i>
Low	0.005 – 0.010
Medium	0.010 – 0.025
High	0.025 – 0.050
Very high	0.050 – 0.100
<i>Degree of meandering</i>	<i>m</i>
Minor	1.000
Appreciable	1.150
Severe	1.300

(1998) and Sand-Jensen and Pedersen (1999). They discovered that plants cause a disruption of the expected logarithmic vertical velocity profile, both within and above plant beds. Measurements of velocity at incremental points moving upwards from the channel bed showed that the velocity within vegetation beds remained at a fairly constant level when measured at locations of progressively decreasing depth while a rapid increase in velocity was observed over a short distance just above the vegetation. An increase in velocity towards the water surface occurred in a logarithmic fashion thereafter.

It could be argued that increases in velocity outside vegetation beds may compensate somewhat for the decrease within plant stands, and that, overall, reach velocity and stage could remain relatively unaffected. This compensatory action may, however, be subject to a maximum biomass limit. It is plausible to suggest that a critical percentage plant cover exists, after which compensatory action is no longer effective, i.e. velocity reductions inside vegetation are greater than the increases created outside the vegetation. The work of Wilcock *et al.* (1999) would appear to support this 'critical biomass' theory. They describe mean summer reach velocities as reduced by 30 % in a vegetated versus plant free channel. The study focused on a 180m reach of very high summer biomass, up to 370g dry weight m⁻², dominated by the problem macrophyte *Egeria densa* (Brazilian waterweed). Sand-Jensen and Mebus (1996) provided a caveat to this, insofar as a definite upper limit to vegetation biomass may exist, since plants are restricted in their lateral expansion by strong shear forces and unfavourable coarse substrates in the flow channels between plant beds. As an example, they cite the submerged macrophyte *Callitriche cophocarpa* (Water starwort) which is thought to be restricted to a maximum of 70% plant cover

in shallow Danish streams (Andersen and Andersen, 1991). Dawson (1976) suggested that the biomass of *Ranunculus peltatus* is self-limiting in an undisturbed state, and Dawson and Kern-Hansen (1978) suggested that a definitive maximum plant biomass had been reached in sections of the Bere stream, Dorset. Whether or not the self-imposed limits on plant biomass are sufficient to keep plant cover below the theorised critical biomass, thus limiting vegetation effects on stage, will be investigated in this chapter and is a central theme of the thesis.

4.2.3 Sediment

Summer plant growth and associated roughness and velocity changes may also introduce seasonal sedimentation changes by promoting sediment storage in summer and erosion in winter, both of which may influence stage. The reduced velocity environment within plants in summer is thought to facilitate increased deposition and retention of fine sediment and leads to both an increase in hydraulic roughness and a local increase in the level of the river bed. For example, Sand-Jensen (1998) refers to a 1.5cm - 11cm increase in bed level under submerged plants in Danish streams. An increase in bed level will have little or no effect on reach-averaged *water depth*, but may cause an increase in river stage if the whole water column is elevated. Sedimentation beneath plants in summer, however, will not cause changes in stage if *mean* bed level does not increase. It could be argued that low-velocity areas of sedimentation, within plant beds, may be offset by erosion in the high-velocity flow channels that exist between vegetation stands. For example, Wolfert *et al.* (2001), describe ‘obstacle bars’, raised hummocks of fine sediment that form in summer beneath submerged vegetation, and ‘chute channels’, concave channels over gravelly

substrate, which form troughs in the high-velocity areas between plant beds (Figure 4.2). The formation of these bedforms is cyclical; they form during summer plant growth but are levelled in winter following the reduction of plant biomass and increases in discharge. The bedforms then reform the following summer when vegetation stands are re-established. Whether the sediment retention beneath vegetation in summer is sufficient to increase mean bed level, and to what extent, will also be a subject of investigation in this chapter.

Field data at a meso- or macroscale are limited, and most quantified studies focus on the microscale (Madsen and Warncke, 1986; Dodds and Biggs, 2002; Machata-Wenniger and Janauer, 1991; Sand Jensen and Mebus, 1996; Sand-Jensen, 1998 and Sand-Jensen and Pedersen, 1999). Larger-scale studies have focused mainly on the effects of vegetation on summary hydrological parameters such as the mean velocity. Recent work carried out under LOCAR include the mapping of sediment deposits in relation to macrophyte cover at the reach scale and investigations as to the constituents of the fine sediment trapped (Cotton *et al.*, 2006, Gurnell *et al.*, 2006, and Wharton *et al.*, 2006).

4.2.4 Previous estimates of the influence of vegetation on stage

Table 4.2 provides a summary of the quantitative results for water depth and stage increases cited in the literature and serves to underline the uncertainty that exists as to the magnitude and nature of vegetation effects on stage and the need for clarification. For example, Dawson (1978) observed a four-fold (0.4 m increase from a winter depth of 0.1 m for equivalent discharges) increase in summer water

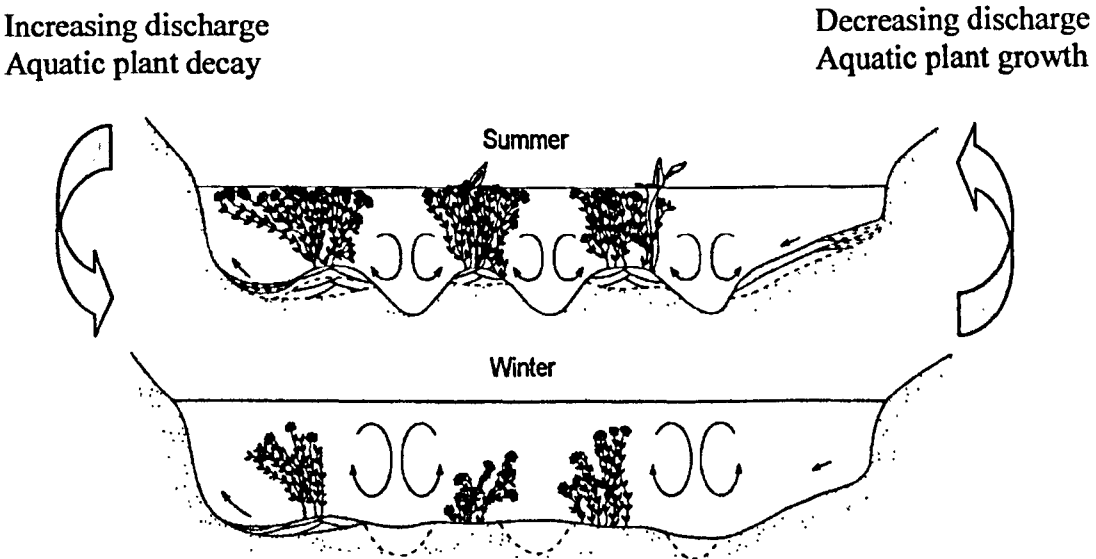


Figure 4.2 Conceptual model illustrating the cyclical bedform changes associated with the growth and decay of aquatic plants (modified from Wolfert *et al.* 2001).

depth compared to the plant free stream, while Wilcock *et al.* (1999) describe a much more modest summer increase in water depth of 40% relative to plant-free conditions. Dawson and Robinson (1984) describe a drop in summer stage level after the removal of plant material in the River Frome of between 0.2 to 0.3 metres and, similarly, Casey and Newton (1973) describe stage level as dropping by 0.3 metres after weed removal. Casey and Newton, however, do not identify the source or context of their estimate. Haslam (1978) estimated that vegetation can typically elevate stage levels by 0.3 – 0.4 m and may cause an increase of as much as 2.0 metres in extreme cases. Again, Haslam does not describe the source or the context of her estimates. Most recently, Naden *et al.* (2006) detailed summer increases in water depth of 0.2 m in a 200 m reach of the River Blackwater dominated by *Sparganium erectum*. They also estimated that plant growth resulted in 50% reduction in velocity. Gurnell and Midgeley (1994) looked at long-term river gauging records on the River Test, Hampshire, and found that annual cycles evident in the discharge record (i.e. high discharge in winter and low discharge in summer) were not mirrored in the stage record, presumably as a result of vegetation influence on stage in summer. Monthly residuals, obtained from a simple linear regression relationship estimated between the stage and discharge data showed a strong cyclical seasonal pattern in mean residual values and scatter, and indicates an underestimation of stage in summer (Figure 4.3). Gurnell and Midegely, however do not offer a quantified depth or stage increase from winter to summer. Several authors cite macrophyte growth as contributing to over bank flooding, for example: Butcher (1933); Hearne and Armitage (1993); Pitlo and Dawson (1990) and Rodwell (1995), but few give quantified evidence or examples. Hearne and Armitage (1993) refer to the River Frome, in 1991, where the highest water levels for that year were

recorded in June. Other than the timing of the flood, at maximum plant biomass, no evidence for macrophytes as a major causative factor is given. Watson (1987) portrays a much more complex set of interactions between vegetation and discharge and describes non-linearity in the summer roughness/discharge relationship, and differences in this relationship depending on the plant species present. No references to the effects of bed level change on river stage were found in the current literature. The comparison of previous results is also complicated by the inconsistent methods of reporting, i.e. either percentage change of depth or change in metres.

4.3 Seasonal patterns and the influence of discharge

The majority of the previous investigations outlined above were conducted in the summer period, and few describe vegetation effects on water level in other seasons. In temperate climates, peak vegetation biomass is generally attained during the summer months (Dawson, 1978; Haslam, 1987; Sand-Jensen *et al.*, 1989; Westlake, 1975; Wilcock *et al.*, 1999), when discharges are usually very low. In this situation, the increase in stage which may occur due to plant growth is less likely to induce over-bank flooding (Champion and Tanner, 2000). Indeed, small increases in stage may be essential to maintain various ecosystem functions (Allan, 1995), or could be manipulated to allow greater water abstraction; while there is no difference in the water available in the river, water level rather than water volume is often an important factor in aquatic ecology (Hearne and Armitage, 1993). If overbank flooding is the most oft-quoted reason for the cutting of aquatic vegetation (Holmes, 1999), and if the impact of macrophyte growth is judged against this datum, there

Table 4.2 Quantitative estimates of the influence of vegetation on water depth and river stage. Estimates regarding water depth will give information on the effect of vegetation induced velocity reduction, while estimates of river stage evaluate the combined influences of vegetation roughness and seasonal changes in mean bed level.

Author	River name	Water depth or river stage	Estimated increase in either depth or stage induced by vegetation
Casey and Newton, 1973	River Frome, Dorset	River depth	0.3 m
Dawson, 1978	The Bere Stream, (Tributary of River Piddle), Dorset	River stage	0.4 m or 400 %
Dawson & Robinson, 1984	River Frome, Dorset	River depth	0.2 – 0.3 m
Haslam, 1978	No details given	River depth	Typically 0.3 – 0.4 m, 2.0 m in extreme cases
Naden <i>et al.</i> , 2006	River Blackwater, Hampshire	River depth	0.2 m
Wilcock <i>et al.</i> 1999	Whakapipi Stream, New Zealand	River depth	40%

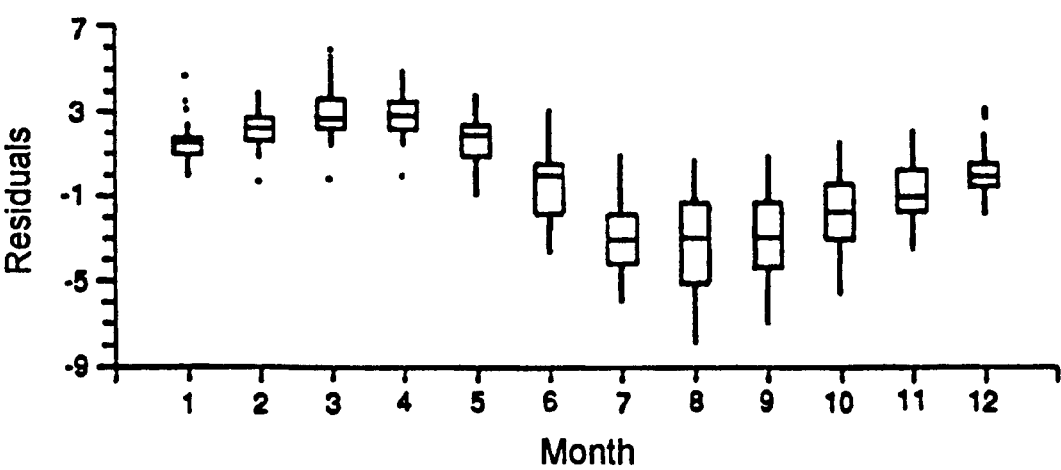


Figure 4.3 Box and whisker plot of monthly residuals, obtained from a simple linear regression relationship estimated between stage and discharge for a 25-year dataset for the River Test, Hampshire. This shows a strong cyclical seasonal pattern in mean residual values and scatter, and indicates an underestimation of stage in summer (Gurnell and Midgley, 1994).

may be a possibility that vegetation management is unnecessary. However, in terms of agriculture, concern may not focus purely on over-bank incidences of flooding. Elevated stage levels in the river will also lead to higher water tables in surrounding fields which can hinder crop production (Haslam, 1978). Therefore, the datum by which to judge the economic effects of vegetation growth cannot be based purely on the extent of over-bank flooding but also on the effect on within bank river stage.

In winter, when discharge is high and flooding is more likely, most macrophyte biomass has been physically removed during flood flows, and most plants either die back to their roots or survive in a much reduced winter growth form (Westlake, 1973; Westlake, 1975; Rodwell, 1995). For example, Flynn *et al.* (2002) describe winter vegetation cover in the River Kennet, Berkshire, as 'negligible', while Dawson (1978) describes typical winter biomass in the River Piddle as being less than 1% of that present in summer. Vegetation effects during winter flows can perhaps be deemed insignificant, and could be said to contribute little to winter flooding given the magnitudes of flow. However, high discharges may occasionally occur during summer (Hearne and Armitage, 1991) and are quite likely to occur at intermediate plant biomass in spring and autumn. In these cases, two mechanisms may act to temper the effects of macrophytes on stage: (i) the reconfiguration of plant morphology; and (ii) the loss of plant biomass due to increased drag and mechanical stress.

4.3.1 Reconfiguration of macrophyte orientation and shape

The effects of macrophytes on stage are not thought to increase linearly with increasing discharge (Watson, 1987). Vegetation effects are believed to be most apparent at low flow and have less effect under high discharge (Sand-Jensen *et al.*, 1989) and this suggests a change in the behaviour of the vegetation. At high discharge, most macrophytic plants (except the most rigid of emergents) tend to change their shape and become more streamlined (Sand-Jensen, 2003). Macrophytes bend in the direction of water flow (Watson, 1987; Pitlo and Dawson, 1990; Wilcock *et al.*, 1999; Champion and Tanner, 2000; Stephan and Gutknecht, 2002; Sand-Jensen, 2003) and constrict in size as their mass is pushed closer together (Dodds and Biggs, 2002; Sand-Jensen, 2003) which greatly reduces roughness (Wilcock *et al.*, 1999).

Sand-Jensen (2003) suggested that the bending of vegetation and greater water depth at high discharge allows a greater proportion of water to pass unimpeded above the vegetation. Similarly, Champion and Tanner (2000) describe differences between shallow and deep sections of a river at the same discharge. They noticed that velocity reductions were much less pronounced in deeper cross-sections than in shallow ones, and that this could not be accounted for by differences in plant cover alone. This may imply that velocity reduction was less in deeper transects because there was more free space available above the vegetation for water to flow. Thus, at high flows, large volumes of water may flow unhindered above the flattened vegetation. Submerged plants in particular change their shape markedly in high flows, and at extreme discharges, plants may bend to the extent that the plants lie flat

against the river bed forming a relatively smooth layer for the water to flow above. In this way, and in comparison to the bed substratum in the absence of vegetation, roughness may conceivably be reduced in the presence of submerged macrophytes. In this circumstance, river stage could even be reduced at high discharges in a densely vegetated stream relative to its plant-free state.

Wilcock *et al.* (1999) postulate that a ‘critical flow’ exists in each reach, after which macrophytes become overwhelmed. They suggested a critical flow of $0.23 \text{ m}^3 \text{ s}^{-1}$ for their small study reach, after which, mean reach velocities were observed to increase by 60%. Similarly, Watson (1987) describes three roughness regimes based on n-VR relationships: (i) an upper roughness regime occurring at low discharge where vegetation controls water flow; (ii) a transition regime where vegetation begins to be ‘drowned out’; and (iii) a lower roughness regime where vegetation effects are negligible (see Figure 4.4). The n-VR curve was originally developed from experiments in grass-lined drainage channels but Watson extended their use, and obtained similar results, in flume experiments and field measurements of chalk stream plants, including *Ranunculus* spp. Chow (1959) presents several n-VR curves for different species of grass and suggests that each species has a unique n-VR curve which is dictated by the flexibility of the vegetation (Figure 4.5).

4.3.2 Reduction of biomass

At high discharges, the effective biomass and roughness of macrophytes is further reduced by the removal of plant parts through mechanical stress (Dawson, 1976; Ham *et al.*, 1981; Pitlo and Dawson, 1990; Wilcock *et al.*, 1999; Flynn *et al.*, 2002)

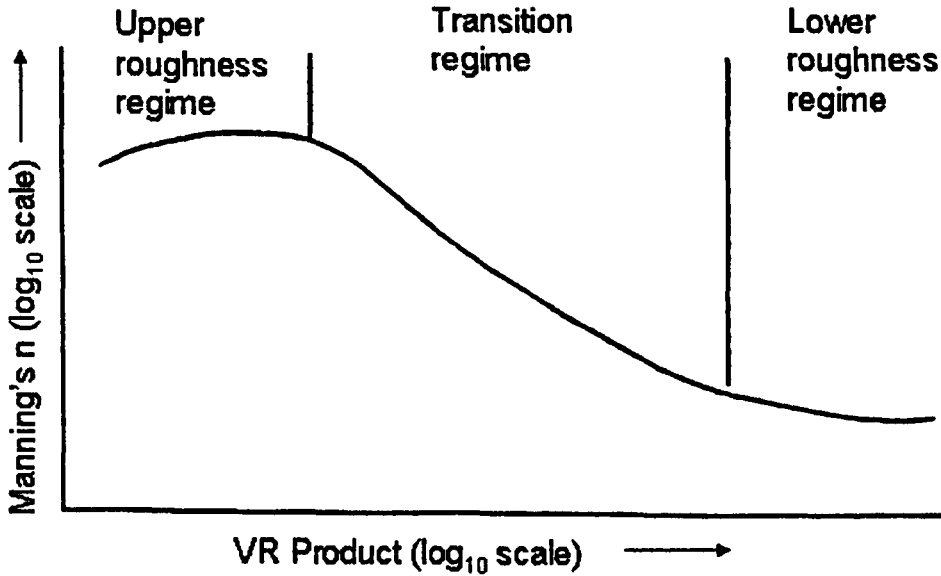


Figure 4.4 Generalised relationship between Manning's n and the product of velocity and the hydraulic radius (VR), used as a surrogate for discharge. The figure is based on results in grass-lined channels of constant biomass and shows the decrease in vegetated roughness with increasing discharge (Watson, 1987).

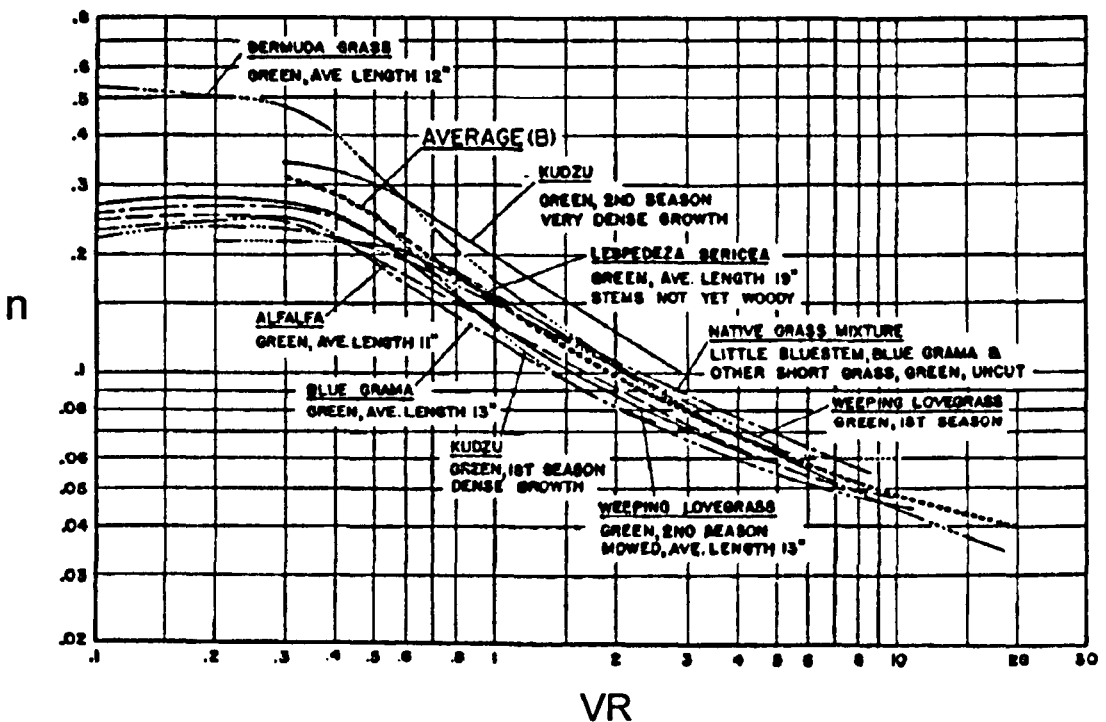


Figure 4.5 n -VR curves for several species of grass in grass-lined test channels which suggest that each species has a unique n -VR curve dictated by the flexibility of the vegetation (Chow, 1959).

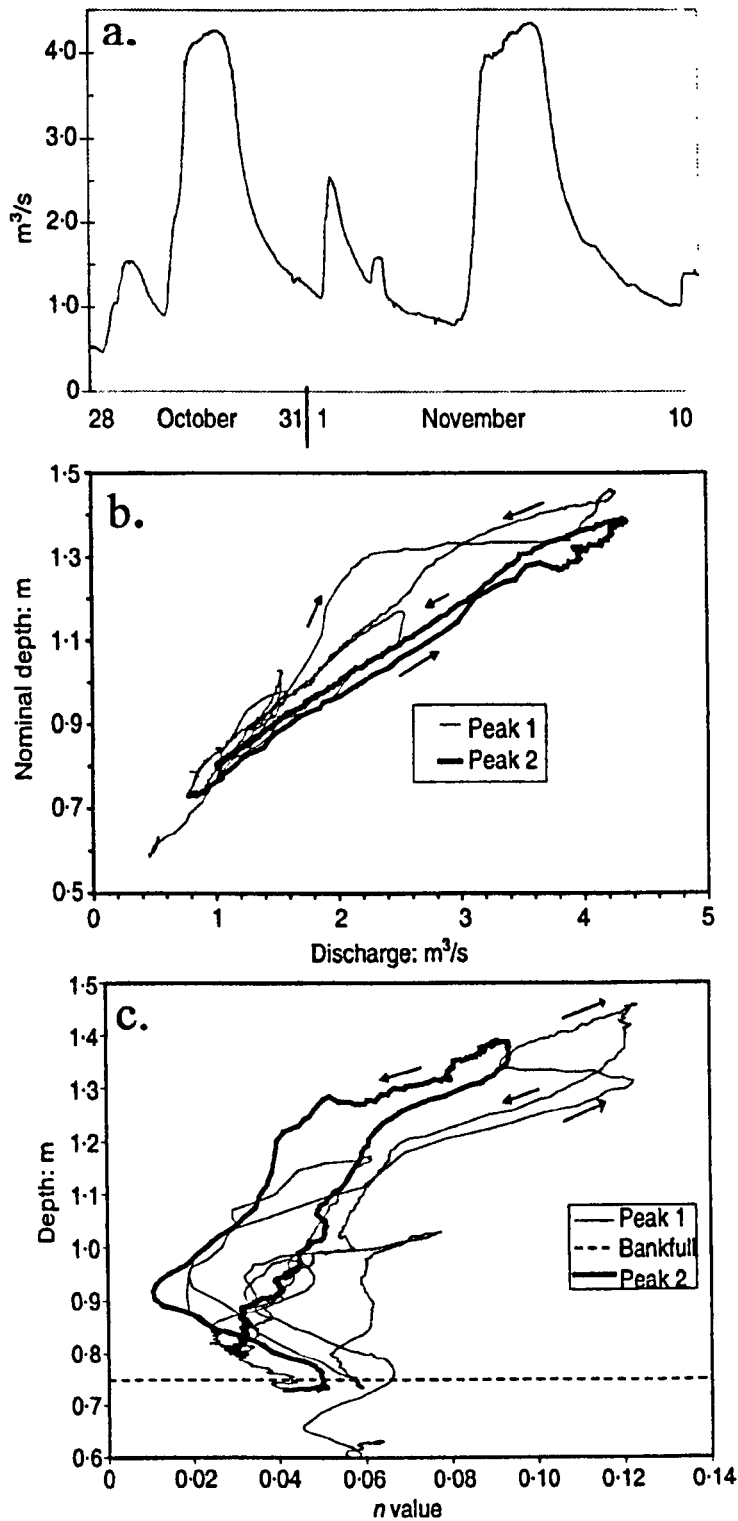


Figure 4.6 a) discharge hydrograph for two consecutive flood events on the River Blackwater, 28th Oct – 10th Nov 2000, b) the corresponding stage/discharge curve showing increased conveyance in the channel for the second flow event and on the falling limbs of each event and c) related n values showing a lowering of the roughness coefficient for the second flood event (Sellin and van Beeston, 2003).

or the washout of whole plants (Pitlo and Dawson, 1990). Removal of vegetation in this way should reduce roughness, increase velocity, and thereby reduce stage. Sellin and van Beeston (2003), describe two hysteresis effects observed in the stage/discharge curves of high-flow events on the River Blackwater, Hampshire (Figure 4.6). They studied two consecutive autumn flood events, of similar discharge, and found that the conveyance capacity of the channel was greater during the second flood event than for the first. This corresponded to a decrease in roughness from the first event to the second. They also found differences within the individual flood events, whereby conveyance capacity was higher, and roughness was reduced, on the falling limb of the hydrograph when compared to values for the rising limb. They attributed both these phenomena to the flattening of live vegetation and the removal of dead plant material under high flows. In chalk streams, summer discharges are generally not sufficient to rip-out healthy plants (Ham *et al.* 1981), and this process is perhaps more applicable to high spring flows when shoots are immature (Haslam, 1978) and autumn high flows just prior to senescence of the plants (Westlake, 1973; 1975). Plates 4.1a and 4.1b provide primary evidence of plant washout on the River Frome: *Ranunculus* debris attached to overhanging tree limbs following flood events in April and September 2004.

4.4 Spatial and temporal scales covered by the hydraulics field data

This reach-scale hydraulics chapter utilises two main elements of the field data: (i) long-term automated pressure transducer (PT) records of river stage, taken at 15-minute intervals; and (ii) multiple discharge measurements, taken at regular intervals throughout the time period covered by the PT records but with a much more



Plate 4.1 *Ranunculus* debris caught by overhanging branches on the River Frome at Hydeclift Plantation in a) April 2004 and b) September 2004.

restricted frequency. Figure 4.7 displays the timing of discharge measurements at the Crockways and Frampton Estate field sites and the relative length of each PT record.

4.4.1 Pressure transducer records

The operation and installation of the pressure transducers has been described in detail in Chapter 2. In total, six pressure transducers were installed on the Frome, three each at Crockways and at the Frampton Estate (Figure 2.9, Chapter 2). The pressure transducers were installed along one bank at appropriate, and roughly equidistant, locations in each study reach. The three PTs at Crockways were operational from April 2003 – April 2005, while the three PTs at Frampton were installed slightly later, in May 2003, but similarly collected data until April 2005. After April 2005, only one PT (PT1 at Crockways and PT2 at Frampton) was maintained at each site to act as a reference to the discharge measurements and these remained operational until October 2005. This provides over two years of continuous stage data. In terms of PT nomenclature, each PT is named first for its site location, either FR1 (Crockways) or FR2 (Frampton Estate), and secondly for its position along the length of the reach, running PT1 - PT2 - PT3 from upstream to downstream. Data control issues are dealt with in Chapter 2.

4.4.2 Discharge measurements

In total, 98 discharge measurements are considered: 49 at each of the two main discharge cross-sections at Crockways and Frampton estate. At Crockways, initial

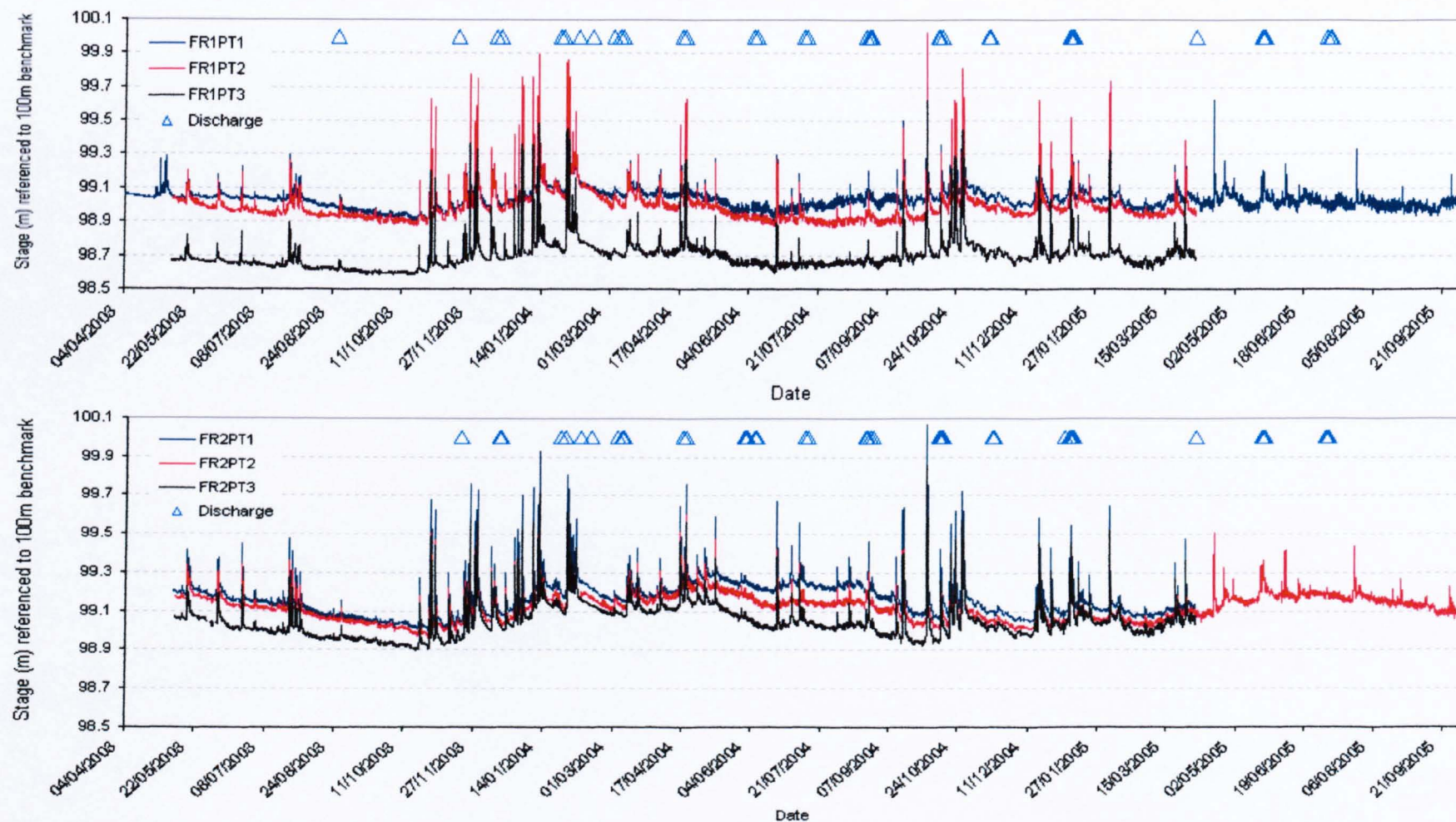


Figure 4.7 PT records of stage and associated discharge measurements at a) Crockways and b) Frampton Estate, April 2003 – April 2005.

discharge measurements were based at both PT1 and at PT3. PT3 is a shallow cross-section ideal for winter measurements but less suitable in summer and a decision was made to combine the discharge measurements at PT3 and PT1, and from April 2004 onwards the discharge measurements continued at PT1 only. The discharge measurements at the PT1 and PT3 cross-sections were both referenced to stage data from PT1. The chosen cross-section at Frampton for all discharge measurements was at PT2.

4.5 Seasonal changes in the stage/discharge relationship

Initial analyses focused on examining the relationship between stage and discharge at each field site and in each season to provide a broad indicator of vegetation influence which could later be unravelled by examining contributing parameters in more detail. Time-linked, stage and discharge data were plotted in a scatter graph, to which simple linear regressions were applied. The linear regression equations provide stage/discharge ratings, which help to describe the stage/discharge relationship quantitatively. Stage/discharge ratings are unique to each channel cross-section and will change along a river reach according to differences in cross-section shape and local slope and the affect these factors have on hydraulic roughness. Similarly, the temporal changes in cross-section characteristics and roughness imposed by vegetation cycles will create change in the stage/discharge relationship and in the rating at a single cross-section. In an extensively vegetated river such as the Frome, it is anticipated that the stage/discharge relationship will change from summer to winter as vegetation cover increases, necessitating perhaps several stage/discharge

ratings. It is predicted that a given discharge in summer will produce a higher stage than will the equivalent discharge in winter.

Figure 4.8 and Figure 4.9 show the stage/discharge graphs for both the Crockways and the Frampton Estate field sites. The data at each site have been subdivided into summer and winter groups based on observed differences in the location of summer and winter data points, and with reference to the expected timing of vegetation influence as espoused in the literature. The regression relationships are strong for each site in each season: summer at Crockways R^2 ($P < 0.00$); winter at Crockways R^2 ($P < 0.00$); summer at Frampton R^2 ($P < 0.00$) and winter at Frampton R^2 ($P < 0.00$) and the discharge data do not appear to show any transitional observations linking the summer and winter data bodies and the switch between summer and winter states appears to be very rapid. The data available for spring and winter is probably of insufficient temporal detail to pick up the transition period but it is thought that two short transition periods may be experienced in the river, during early April and in either September or early October (See Table 4.3). The rapid transition between winter and summer states points to the possible existence of a critical biomass, which must be attained before vegetation has any effect on river stage.

4.5.1 Estimates of seasonal stage increases: high and low stage

Several immediate contrasts in the stage/discharge relationships can be observed between seasons and between sites. At Frampton, there appears to be a distinct difference in the stage/discharge relationships observed in summer compared to

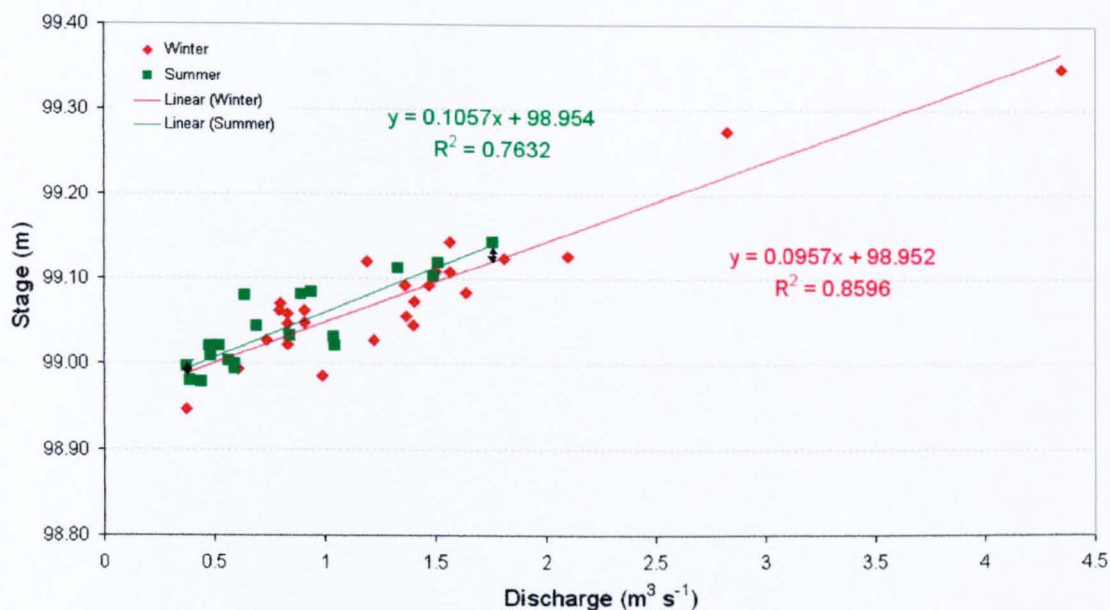


Figure 4.8 Discharge measurements at Crockways with reference to stage at FR1PT1 (August 2003 – July 2005). The data have been divided into summer and winter groupings with reference to the expected growth cycle of aquatic plants and to differences in the plot locations of individual data points.

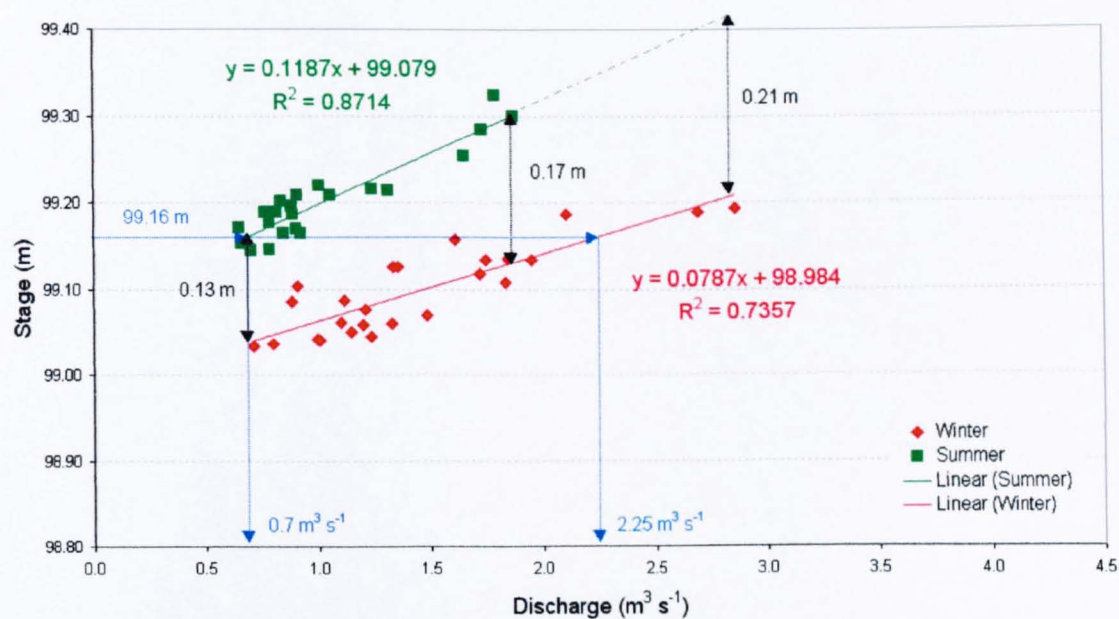


Figure 4.9 Discharge measurements at Frampton Estate with reference to stage at FR2PT2 (October 2003 – July 2005). The data have been divided into summer and winter groupings with reference to the expected growth cycle of aquatic plants and to differences in the plot locations of individual data points.

Table 4.3 Discharge dates categorised as ‘winter’ and ‘summer’ for the water years 2003/2004 and 2004/2005. The transition period between winter and summer has been estimated by combining dates from both water years.

Water year	‘Winter’	‘Summer’
2003/2004	20/11/2003 – 10/03/2004	19/04/2004 – 28/08/2004
2004/2005	12/10/2004 – 06/04/2005	21/05/2005 – 07/07/2005
Water years combined (2003 – 2005)	12/10 – 06/04	19/04 – 28/08
Estimated transition period	28/08 – 12/10	06/04 – 19/04

winter, whereas at Crockways the data do not suggest that there are any significant seasonal differences in the stage/discharge relationship. On a very simple level, the summer data body at Frampton plots above the winter observations and there is no overlap between the summer and winter data bodies. This indicates that stage levels are correspondingly higher in summer than in winter at similar discharges. In contrast, the Crockways summer and winter data bodies appear less segregated: the summer data points generally plot slightly higher than the winter observations but several points from each group may be seen to overlap. The difference between the summer and winter stage values can be examined using the linear regression lines which provide an average value for river stage for a given discharge. If the summer and winter regression lines at each site are compared, then, on average, and within the range of the available data, the net increase in stage at Frampton in summer, with reference to the winter baseline, is 0.12 – 0.17 m (Figure 4.9). The Crockways values, in contrast, demonstrate a much smaller average net increase in stage of 0.01 – 0.02 m from the winter baseline, at equivalent discharges and within the range of the data available. If the linear regressions are assumed to hold true for a wider range of discharges, and if the summer regression lines are *tentatively* extended to cover the same discharge range as covered in winter, then the winter and summer stage differences become much more marked, reaching 0.045 m at Crockways and 0.21 m at Frampton.

These various discharge-defined stage increases indicate that the summer and winter regression lines are not parallel, and that differences in slope exist between seasons. This can be quantified by comparing the coefficient of each regression equation: Crockways has a winter regression multiplier of 0.097 compared to a slightly higher

multiplier in summer of 0.10, and, similarly, at Frampton the winter multiplier of 0.08 is much lower than the summer multiplier of 0.12. The higher regression slope in summer at Crockways and Frampton indicates that stage increases with discharge at a higher rate in summer than for the winter baseline and suggests that, correspondingly, vegetation effects on stage are greater at higher discharges. However, the variable effect of vegetation on stage with different discharges may be measured in two ways: *net* seasonal change in stage for a given discharge (the difference between winter and summer stage), as given in Section 4.5.1, and *proportional* changes (the net change considered as a percentage of winter stage). The net stage changes are of more immediate relevance to river managers and will help to predict whether a critical stage will be achieved in summer, i.e. whether flooding will occur, while the proportional measures perhaps reveal more about hydraulic resistance effects. Both measures were calculated for the Frampton site.

When considering proportional stage changes it is important to remove the 100m reference built in to the PT measurements (stage values were measured with reference to an arbitrary fixed position) because percentage measures will be sensitive to the winter baseline value. This recalibration was achieved by using the channel bed level beneath the PT, as measured during initial topographic surveys (May 2003), as an approximation of bed level and referencing changes in stage to this baseline level. Using these recalibrated stage measurements, the percentage increase at the lower discharge estimate is 26.6 %, while at the higher discharge estimate, the summer stage is higher by 30.1 %. This indicates that, even in proportional stage terms, vegetation influence is higher at higher discharge.

Stage increases may also be considered in terms of discharge by comparing the winter and summer discharges which would be required to obtain the same stage value. For example, at the lowest summer stage, 99.16 m, a summer discharge of $0.7 \text{ m}^3 \text{ s}^{-1}$ (6.4 % of winter bankfull discharge) will correspond to the same stage level as a winter discharge of $2.25 \text{ m}^3 \text{ s}^{-1}$ (20.5 % of winter bankfull discharge), despite the summer discharge being 69 % lower than the winter discharge (see Figure 4.9). At the highest recorded summer stage, 99.3 m, a summer discharge of $1.9 \text{ m}^3 \text{ s}^{-1}$ (17.4 % of winter bankfull discharge) will correspond to the same stage level as a winter discharge of $4.05 \text{ m}^3 \text{ s}^{-1}$ (37.0 % of winter bankfull discharge), despite the summer discharge being 48 % lower than the winter discharge. These figures would seem to suggest that vegetation influence is *less* at higher discharges than at low discharges. This alternative ‘equivalent discharge’ measure of vegetation influence at different discharges suggests that some conflict between the current and previous research may arise due to the method of comparison used to describe the vegetation effect with changing discharge. Three methods of comparison are presented here, but it is the net stage increase that is most meaningful, particularly for river managers.

The trends in net and proportional stage increases, which suggest vegetation influence is greatest at high stage, are in opposition to results reported by Watson (1987) for the River Ebble. Watson suggested that the effect of vegetation on the discharge/roughness relationship is non-linear, and that vegetation effects on roughness diminish with increasing discharge beyond a critical flow limit (Figure 4.4). However the stage/discharge measurements may not be strictly comparable, for instance the range of discharges covered by the summer data in this study may not be comparable to the range for the River Ebble: Watson suggested that stage at first

increases with increasing discharge and later decreases, and, therefore, perhaps only the early part of this trend is represented here. Watson lists the discharge range for one case study site as $0.4 - 1.5 \text{ m}^3 \text{ s}^{-1}$ which is very similar to the sampled summer discharge range at the present study sites: $0.37 - 1.76 \text{ m}^3 \text{ s}^{-1}$ at Crockways and $0.65 - 1.87 \text{ m}^3 \text{ s}^{-1}$ (6.3 % - 30 % of winter bankfull discharge) at Frampton (5.9 % - 17.1 % of winter bankfull discharge) but Watson does not provide a bankfull or median bankfull reference for his discharges and, therefore, it is difficult to compare the two analyses. The review of previous research in Chapter 2 indicated that summer discharges in the Frome, in common with most chalk rivers, are very stable and that very few storm events are recorded in the summer season. This is confirmed in Figure 4.6, which illustrates the dates of discharge measurements in relation to flow conditions and indicates that the sampled discharges cover the range of conditions experienced on the Frome in summer. In this context, assuming an adequate coverage of possible discharges, Watson's theory of diminishing effects with discharge is not applicable on the River Frome and, instead, the effects of vegetation are increased at higher summer stages.

Discharges in the Frome may not be sufficient to cause the compression of submerged plants and biomass loss, or perhaps the increased roughness generated by the submergence of *emergent* plants is greater than the decrease in roughness expected following the compaction of submerged vegetation. Watson suggested that emergent plants may *increase* channel roughness with increasing discharge and that the n-VR curve is only strictly applicable to submergent plants. Chow (1959) suggested that n-VR curves are unique to each species and it may be that the results

evident here represent the combination of submergent and emergent species with very different 'flexing' limits.

4.5.3 Site differences

The conspicuous site differences displayed between Crockways and Frampton are somewhat unexpected. It was thought that a similar, and significant, increase in summer stage, relative to discharge, was likely to occur at both sites as they are situated so close together and both have relatively high vegetation abundance. As a check against the possible local influence of channel cross-section, and/or, the possibility of errors in the primary PT data, the original Crockways and Frampton discharge data were compared to stage values from the other two PTs at each site (Figure 4.10 a – 4.10 d). It is evident from these plots that the site differences are replicated at all three pressure transducers within each site and that differences would be equally marked whichever combination of PT data sets were chosen. Thus, an alternative, physical, explanation must be sought.

The Crockways and Frampton field sites differ in two major ways: in (i) the complexity of local channel morphology and relative water depth; and (ii) the abundance of macrophytic vegetation. The Crockways site is of complex channel morphology: the planform is highly sinuous and the in-channel features are diverse, including several very deep pools, two small islands, and two riffles (see Figure 2.9 and Plates 2.3 and 2.6). This in-channel diversity leads to patchiness in vegetation cover, and macrophytes were conspicuously absent in deep pools and fast-flowing riffles. Emergent macrophytes are much more restricted in their distribution at

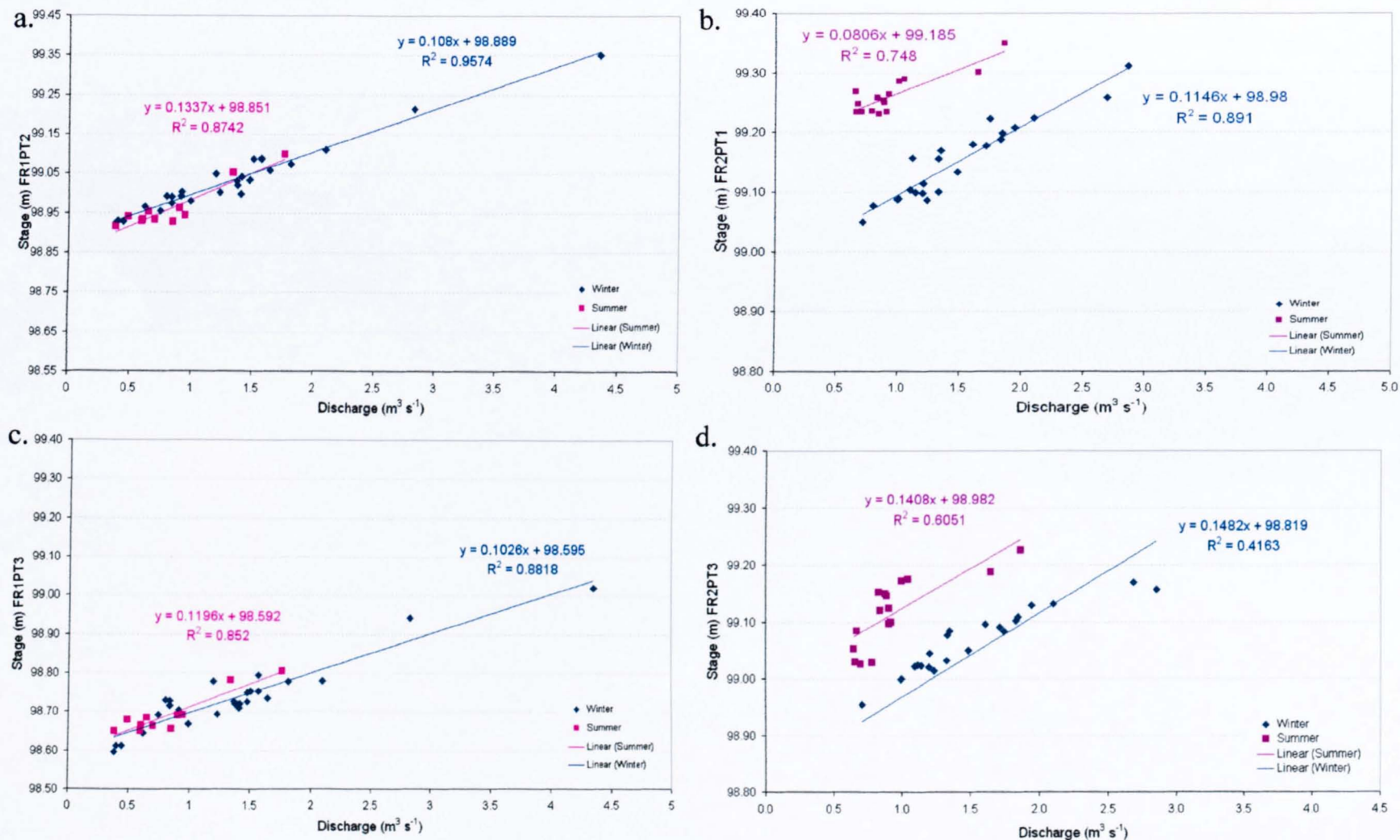


Figure 4.10 Discharge measurements at Crockways and Frampton with reference to: a). FR1PT2; b). FR1PT3; c). FR2PT1; and d). FR2PT3.

Crockways than at Frampton, and are almost unrepresented in the river section covered by the PTs. The water within the PT reach would appear to be either too deep in the pools and too fast flowing in the riffles to allow the growth of emergent plants. Side bars are also absent from the PT reach and this further restricts opportunities for emergent growth. In the straight, and shallow, sections upstream and downstream of the PT reach, emergent and riparian species are both represented and the plants greatly increase in abundance. However, even here, the plants are not quite as abundant as in the Frampton reach and individual plant stands do not tend to be as large at Crockways as those at Frampton. It could be theorised that a critical plant cover, as discussed in Section 4.2.1, has been met at Frampton and not at Crockways, i.e. the plant cover present at Frampton is sufficiently abundant to mean that decreases in velocity within plant beds may no longer be compensated for by increases in velocity in un-vegetated areas; while at Crockways, sufficient un-vegetated space remains to allow compensatory increases in velocity outside plant beds to counteract velocity reductions within vegetation stands. The relative importance of vegetated and un-vegetated space in influencing mean reach velocity is considered in more detail in Chapter 5.

The reasons for the marked differences evident between the two study sites may also be a guide to interpreting the very different estimates of vegetation influence on stage and water depth published in the literature (Table 4.2). The results from Crockways and Frampton are much lower than all of the estimates quoted in the literature: Crockways varies from 0 – 0.02 m and Frampton varies from 0.12 – 0.17 m within the range of the available summer discharge data, and 0.21 m if the regression relationship at Frampton is tentatively extended to cover the winter discharge range.

Mann-Whitney tests on the summer and winter stage data for each site suggest that only the Frampton increase in summer stage is statistically significant (Crockways $P = 0.792$; Frampton $P = 0.00$). A non-parametric test was chosen in preference to student's t-test as the basic assumption of data normality was not met. The estimates of Dawson and Robinson (1984) and Naden *et al.* (2006) are closest to the Frampton results, but the *highest* value at Frampton is at the *lower* boundary of their estimates. Compared to the extreme end of the stage increase estimates, a 2.0 m increase in extreme cases (Haslam, 1978), the Frampton results are lower by a factor of 10. However, these estimates come from several different river sites in the UK, and each site will have unique in-channel morphology, vegetation abundance, vegetation assemblages and sediment supply which will affect vegetation influence on stage.

Vegetation abundance is the most obvious factor which will affect the magnitude of seasonal stage increases. It is likely that a critical biomass must be attained before vegetation roughness will affect stage, but it is also likely that the overall magnitude of the stage increase will be affected by vegetation abundance, for example two sites may have attained a critical biomass but a higher vegetation abundance at one site may mean that stage increases will be greater. The RHS analysis in Chapter 3 indicated that the Frampton site has a very high vegetation cover compared to the average vegetation cover for the national chalk group and it was thought likely that the stage effects at this site would be similarly high when related to the results from the literature. However, vegetation abundance is not the only factor affecting the vegetation influence on stage. Watson (1987) suggested that vegetation *type* is an important factor and that rigid emergent vegetation will have a greater impact on hydraulic roughness, and on stage, than submergent vegetation. Therefore, the stage

increase observed at two sites of similar vegetation biomass will differ if the relative abundance of emergent and submergent plants differs.

Site differences in the magnitude of stage increases may also arise from the shape of the channel cross-section; for example, at two sites of similar vegetation influence the stage increase (if measured as vertical change in metres) at a site with a narrow cross-section will be higher than in reaches with a wider cross-section, because the increase in water cross-sectional area induced by plant growth must be accommodated across a wider reach.

The analysis in Chapter 3 revealed that the Frome has a generally higher width/depth ratio than either the average for chalk rivers or for rivers of other geology. This means that stage increases on the River Frome may not appear as significant as on other chalk rivers of a lower width/depth ratio. Seasonal scour and fill of fine sediment is also thought to affect river stage through mean bed level change and this will vary from site-to-site, depending on the abundance of vegetation upstream and the incidence of fresh sediment supply, e.g. from tributary rivers, along the reach. At Frampton, given the high vegetation abundance, sedimentation is likely to be high relative to sites of lower vegetation cover and this will be investigated later in Section 4.8.2. One final factor to consider is the changing vegetation influence at low and high discharges. The present study at Frampton shows that stage increase changes with discharge within the range of $0.65 - 1.87 \text{ m}^3\text{s}^{-1}$. The published studies do not list the discharges over which measurements were made and it must be assumed that they quote stage increases at only one discharge or as an average influence over an unknown range of discharges. As a final consideration, the

methods of measurement may also differ greatly. The stage increases at Frampton refer to average summer and winter data collected over two seasons, and considers the natural cycle of vegetation growth and senescence, whereas studies such as Casey and Newton (1973) and Dawson and Robinson (1984) describe abrupt changes in stage which occur after the cutting of large sections of vegetation. The effects observed after the vegetation cut are likely to be more extreme than those observed over a long period of natural change and the short-term studies are likely to yield higher estimates of stage changes.

The site differences described in this study and the very different estimate of stage increases given in the literature (Table 4.2) highlight the importance of supplying contextual information about a study site alongside any estimation of vegetation effects on reach hydraulics. The authors in Table 4.2 give little indication as to the site context of their estimates, very few references are given in the literature as to the vegetation abundance or vegetation type at the field sites, though high biomass is implied (a critical biomass must have been reached for vegetation effects to occur). Naden *et al.* (2003), give a description of the vegetation in their reach. 62% of the channel was affected by vegetation (both submergent and emergent) at low water and this is similar to the vegetation abundance at the Frampton reach. It is encouraging that the results are relatively similar, as it helps to validate the results of the present study and suggests that there is some consistency in the vegetation influence between similar sites. The cited studies do not provide details of the channel shape in their study reaches or any seasonal sedimentation changes and no references are made to the discharge conditions under which the estimates were made. It is difficult to compare the published results to each other and to the results of this present study.

The site differences and discharge influences exposed by the stage/discharge analysis are considered in later sections of this chapter through examination of the effects of vegetation growth on mean reach velocity, channel roughness, hydraulic geometry and sedimentation. The following sections seek to disentangle the many variables and processes which are included within the lumped discharge parameter using derivative variables: hydraulic roughness; average velocity; width; depth; and cross-sectional area.

4.6 The changing roughness regime

Plots of hydraulic roughness (Manning's n) and discharge are shown in Figures 4.11 and Figure 4.12. Values for both the Darcy-Weisbach and Manning's n roughness coefficients were computed and graphically displayed, but the interpretation in this chapter focuses only on Manning's n . The Darcy-Weisbach data were shown to correspond closely to the Manning's data and, accordingly, the Darcy-Weisbach diagrams are presented in an Appendix B of the thesis and not in the main body of the chapter. The roughness data have again been divided into summer and winter data sets, based on the criteria outlined in Table 4.3, but at the Crockways site, further subdivision was required to separate the discharge measurements taken at PT1 and PT3. Discharge measurements from the two cross-sections may be combined and considered together, but cross-sectional area, velocity and slope (and therefore roughness) must be considered separately. No summer discharge measurements were taken at PT3, and as seasonal comparisons were not possible, it

was decided that values from the PT3 cross-section should not be included in any further analyses.

The first point to note from the roughness plots is the generally high values of the roughness coefficients. These values would be considered high for the River Frome if not for their vegetated context. Figure 4.13 provides a diagrammatic representation of Manning's n values obtained in studies of vegetated rivers placed alongside the roughness values obtained from the present study sites. Maximum and minimum values for the Frome study sites are included (coloured red) to the right of Figure 4.13 and appear to sit well within the values from the literature. In addition, Table 4.4 describes the derivation of Manning's n for the Frome field sites using Cowan's method (see Section 4.2.1). The estimated roughness values are 0.193 for Crockways and 0.1455 at the Frampton Estate site, at the top of the measured range for each site, which provides further verification of the measured results at the study sites. Roughness varies from site-to-site depending on changes in bed material, planform, cross-section shape and slope. It is evident that the Crockways cross-section has a greater base roughness in winter than the Frampton cross-section. This is not surprising given the nature of the two cross-sections: the Crockways cross-section is located just before a sharp meander bend, and an associated pool, and this increases roughness and slows water flow (Dingman, 1984). Also, in general, the Crockways cross-section has a more irregular cross-section shape which may increase roughness compared to the flat bed and vertical banks of the Frampton cross-section. The influence of vegetation in summer is greater at Frampton than at Crockways and roughness values at Frampton reach similar levels to that at Crockways in summer.

4.6.1 Site differences and the influence of discharge

Channel resistance is known to vary with discharge. Dingman (1984) lists three factors that contribute to this: (i) an increase in water depth and a lessening of the effectiveness of roughness elements; (ii) changes in the size, shape and spacing of channel bed forms; and (iii) changes in roughness experienced when flows reach out-of-bank levels. Generally, it is usual for channel resistance to decrease with increasing discharge until discharge reaches overbank levels, after which roughness will increase as water moves on to the flood plain (Dingman, 1984). With the Frampton and Crockways data, only the first factor really applies - the second factor is more restricted to channels dominated by sand-sized substrate and the third factor does not apply because all discharge measurements were taken at within bank flows. At Crockways (Figure 4.11 a), roughness can be observed to decrease with increasing discharge in both summer and winter, though at different rates. Summer roughness values appear to decrease at a faster rate with increasing discharge than they do in winter. There is, however, obvious overlap between some of the summer and winter roughness values at Crockways and it may be that there is no significant division between summer and winter values. It is plausible to suggest that the separation in the plot locations of summer and winter observations is more due to summer and winter differences in discharge than to any seasonal differences in the roughness coefficient. If the summer and winter values are treated together, then only a logarithmic regression may be more appropriate: Figure 4.12a shows the linear relationship observed between Manning's n and discharge after \log_{10} transformation of the data and a strong R^2 of 0.77 ($P < 0.00$) is obtained for the transformed data.

At Frampton the summer and winter data sets are more distinct, with no overlap of data points, and the slopes of the regression relationships are also remarkably different (Figure 4.11b). The summer regression line shows the expected negative relationship between discharge and roughness, while the winter regression does not reveal any significant relationship between roughness and discharge, although a weakly *positive* relationship (R^2 0.03; $P < 0.43$) may be observed. Even when considered as log values (Figure 4.12b), there is no improvement in the relationship (R^2 0.02; $P < 0.50$). The lack of a negative, or even a significant, relationship between roughness and discharge in winter at Frampton may be explained by the nature of the Frampton cross-section: the discharge cross-section lies within a straight and uniform reach, which lacks major in-channel features, and the cross-section itself is very homogeneous with generally uniform depth across the river channel. The major contributor to roughness at Frampton is 'skin resistance': both the bed substrate which is composed of coarse gravel and cobbles and the rough earthen banks. Given the lack of change in n with increasing discharge, it is probable that increasing bank roughness compensates for the decline in effective bed resistance as discharge increases. Dingman (1984), presented a diagram that describes theoretical variations in Manning's n with flow depth for different roughness heights and Frampton in winter conforms to the trends shown for the lower roughness heights (Figure 4.14). As a further point, the relatively high width/depth ratio means that an increase in discharge will produce correspondingly less change in water depth than in a channel with a lower width/depth ratio, and hence the effects on roughness will be reduced.

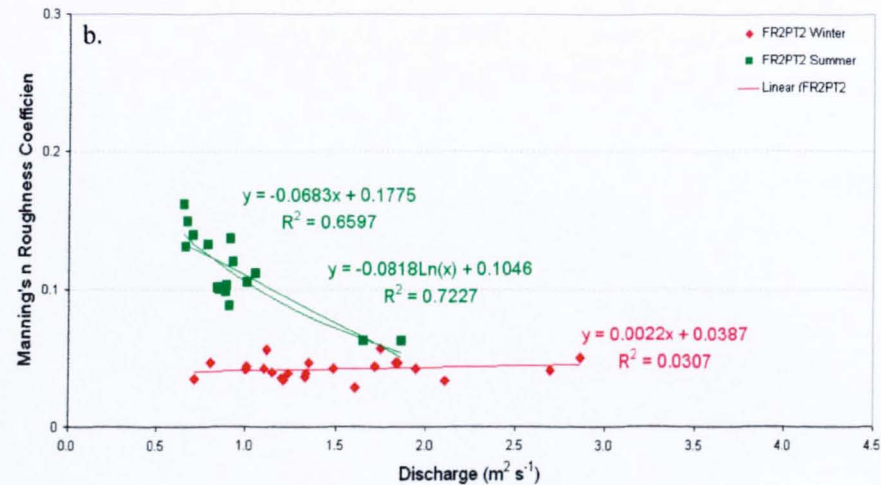
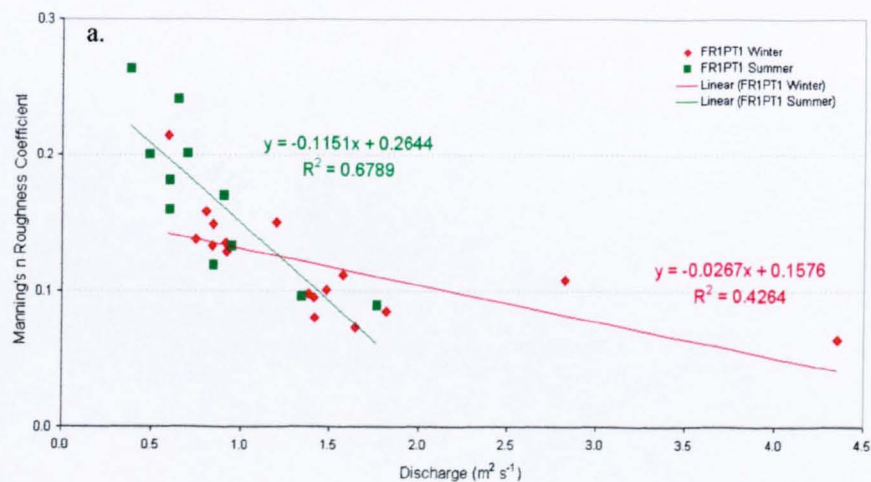


Figure 4.11 Discharge/Manning's n plots for: a) Crockways; and b) Frampton estate, September 2003 – January 2005.

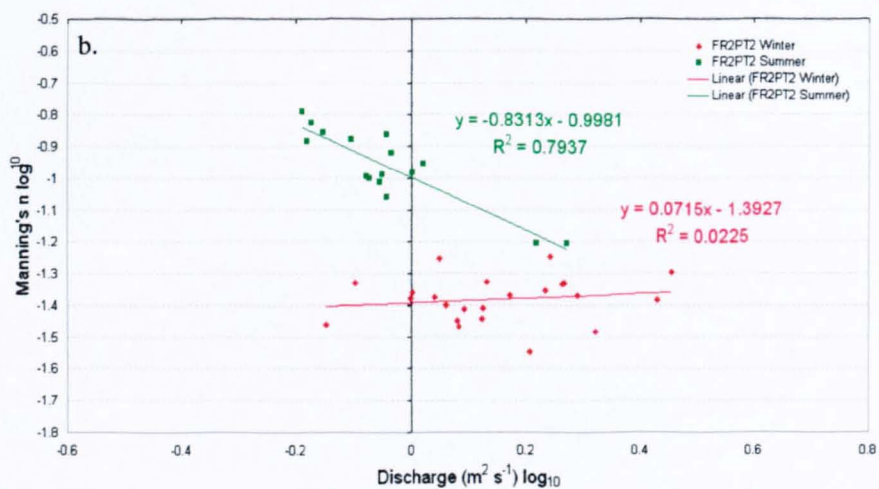
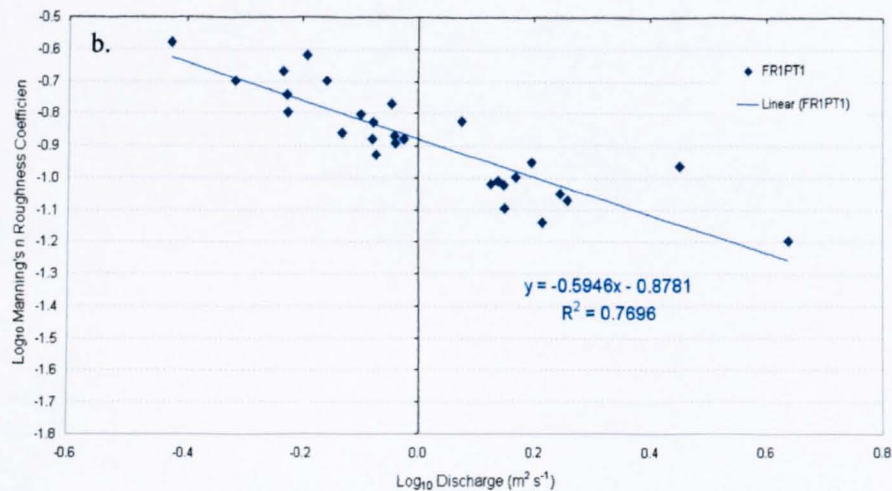


Figure 4.12 Log_{10} Discharge/Manning's n plots for: a) Crockways; and b) Frampton estate, September 2003 – January 2005.

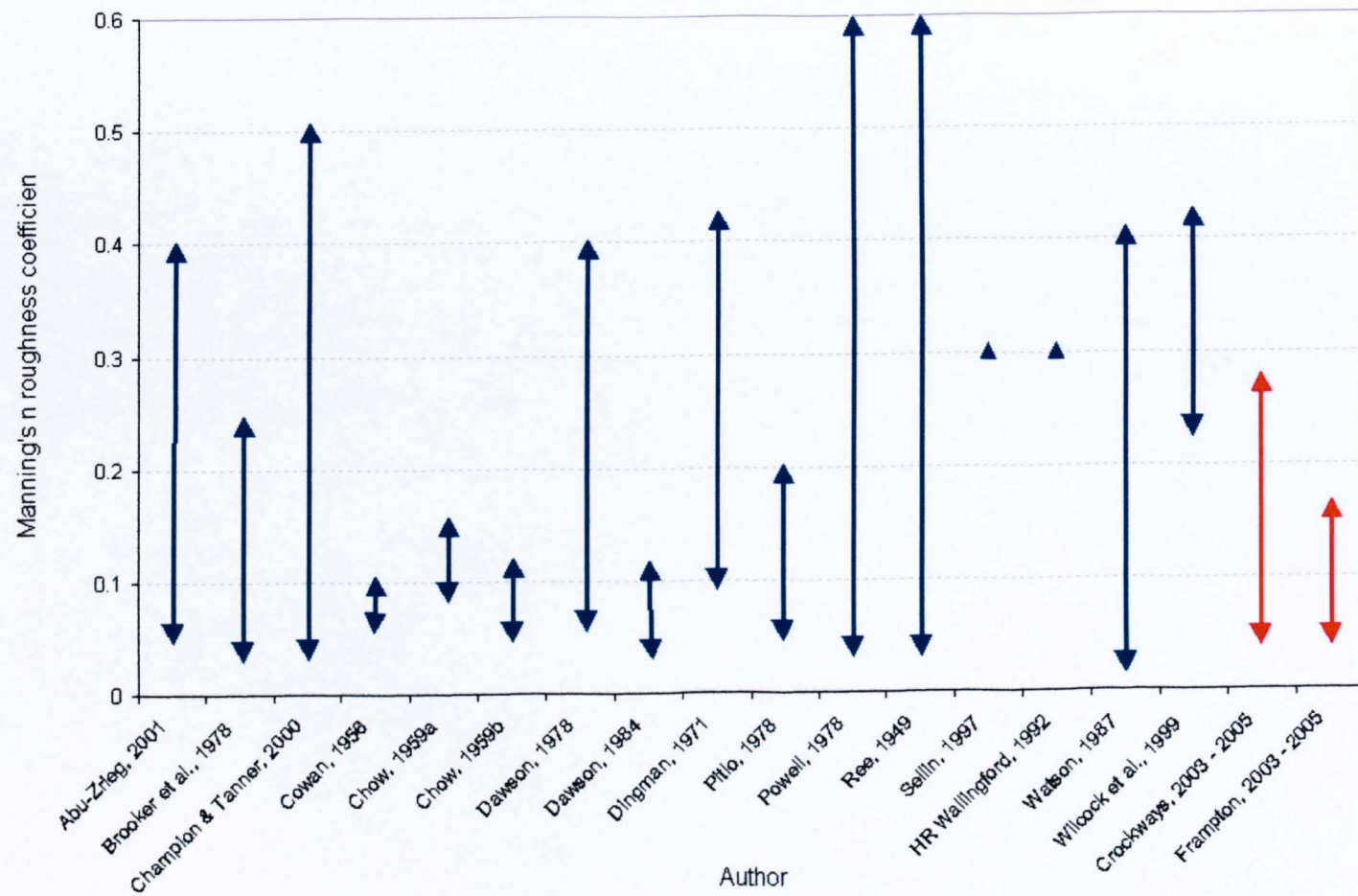


Figure 4.13 Comparisons of maximum and minimum n values obtained by several authors in a vegetated context (Sellin (1997) and HR Wallingford (1992) report only one value of Manning's n), the results from the present study are indicated in red to the right of the diagram and these are shown to be consistent with the range of values listed in previous research..

Table 4.4 Estimated Manning’s n for the Crockways and Frampton Estate field sites in summer, derived using the method of Cowan (1956).

Channel conditions	Crockways	Frampton
Material involved	Coarse gravel: 0.028	Coarse gravel: 0.028
Degree of irregularity	Moderate: 0.01	Minor: 0.005
Variations of channel cross-section	Alternating occasionally: 0.005	Gradual: 0.000
Relative effect of obstructions	Appreciable: 0.020 – 0.030	Minor: 0.010 – 0.015
Vegetation	Very high: 0.005 – 0.1	Very high: 0.05 – 0.1
Degree of meandering	Appreciable: 1.15	Minor: 1.0
Total n	0.193	0.145

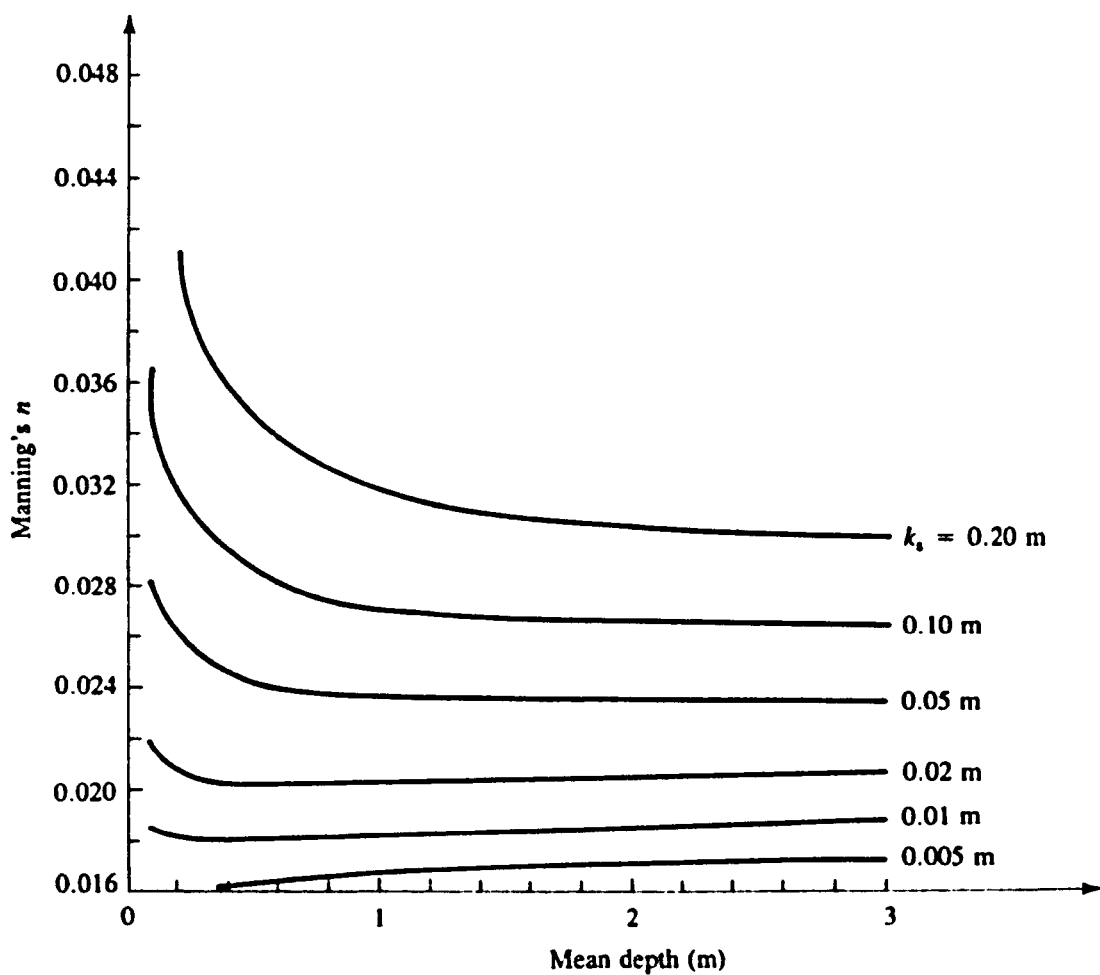


Figure 4.14 Theoretical relationships between increasing mean depth and Manning's n for various roughness heights. The three lower relationships are similar to winter the roughness/discharge relationship at the Frampton Estate field site (From Dingman, 1984).

Summer roughness values at Frampton are much higher than winter values: the mean of n values in summer is 0.11 while the winter mean is 0.04, and, similarly, the Darcy-Weisbach summer mean is 3.27 compared to a winter mean of 1.27 (see Appendix B). Summer roughness also shows a definite negative relationship with discharge compared to the winter situation and indicates that roughness decreases significantly with increasing discharge. If the summer roughness relationship is treated as separate to the winter relationship then they can be plotted on a logarithmic scale (Figure 4.12b), in a similar manner to the combined values at Crockways (Figure 4.12a). The \log_{10} transformation increases the R^2 correlation coefficient for the summer regression relationship from 0.66 ($P < 0.00$) to 0.79 ($P < 0.00$).

Vegetation has a much higher effective roughness height than the bed substrate and a decline in roughness values with discharge in the vegetated summer period may be explained by three linked factors: (i) as described by Dingman (1984), an increase in discharge will lead to a decrease in roughness as the ratio of depth to the height of the roughness elements increases; and this is exaggerated in vegetated streams where (ii) submergent vegetation is compacted and flattened as depth and velocity increase (Watson, 1987 and Sand-Jensen, 2003); and (iii) vegetation biomass may be lost as velocity increases (Dawson, 1976; Ham *et al.*, 1981 and Flynn *et al.*, 2002). The extent of the roughness reduction in summer, with the summer roughness values at high discharge very close to winter values at the same discharge, suggests that vegetation effect on hydraulic roughness is reduced at higher discharges: the data suggest that the plants *do* bend and that biomass *is* lost at high flow. However, these mechanisms for reducing roughness do not appear to have the expected influence on stage: stage increases at a higher rate with increasing discharge in

summer than in winter indicating that the net vegetation effect is greatest at high flow.

The summer decline in roughness at Frampton with increasing discharge, as shown in Figure 4.11a and Figure 4.11b, is not consistent with summer trends observed in the stage data (Figure 4.9). In the stage/discharge relationship, the net stage increase between the winter baseline and summer stage was highest at higher discharges, whereas the reduction in resistance shown here would ordinarily suggest a lessening of the impact of the vegetation in summer at higher discharges. To explore this discrepancy further, stage was compared directly to Manning's n at Crockways and at Frampton (Figure 4.15 and Figure 4.16). This shows the same general patterns as evident in the discharge/Manning's data but with significant differences in the significance of the regression relationships. The summer stage/ n values at Frampton display a moderate negative relationship (R^2 0.53; $P < 0.01$) while the winter values show a weaker positive relationship (R^2 0.27; 0.609) between stage and n . The weak relationships for stage and Manning's n values indicates that not all the variation in stage may be explained by roughness, and that other factors may have an effect on stage.

Two other factors which may influence stage and create scatter include: (i) volumetric displacement of water by plants in ponded river sections; and (ii) changes in mean bed level. In terms of volumetric displacement, Mitchell (1974) refers to results obtained by Westlake (1968) who estimated that plant mass on the River Frome occupied 0.3 % of the channel volume. This evidence would suggest that the influence of volumetric displacement is unlikely to be significant, but Westlake's

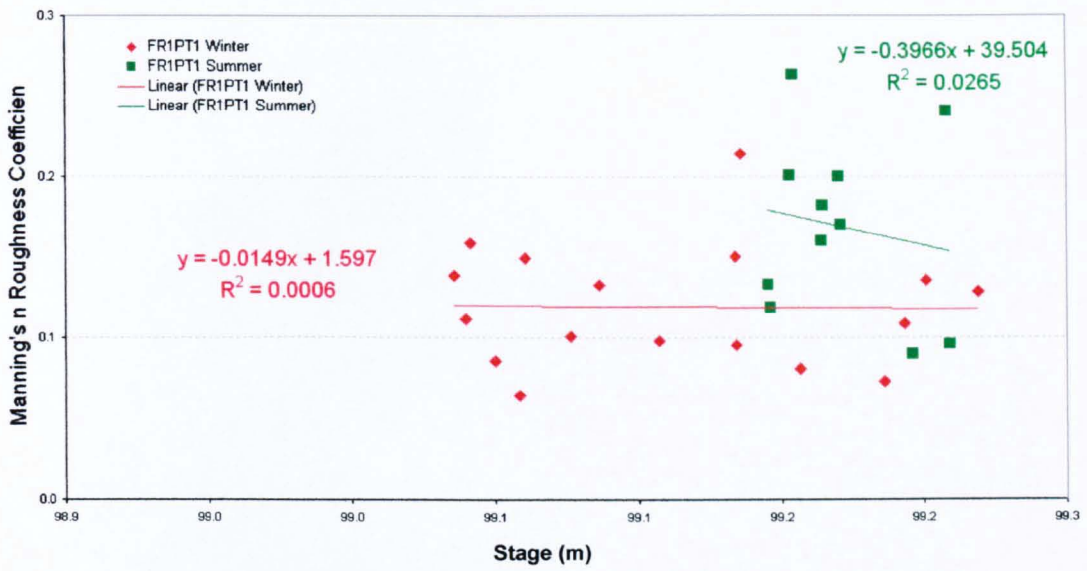


Figure 4.15 Stage/Manning's n at Crockways, September 2003 – January 2005

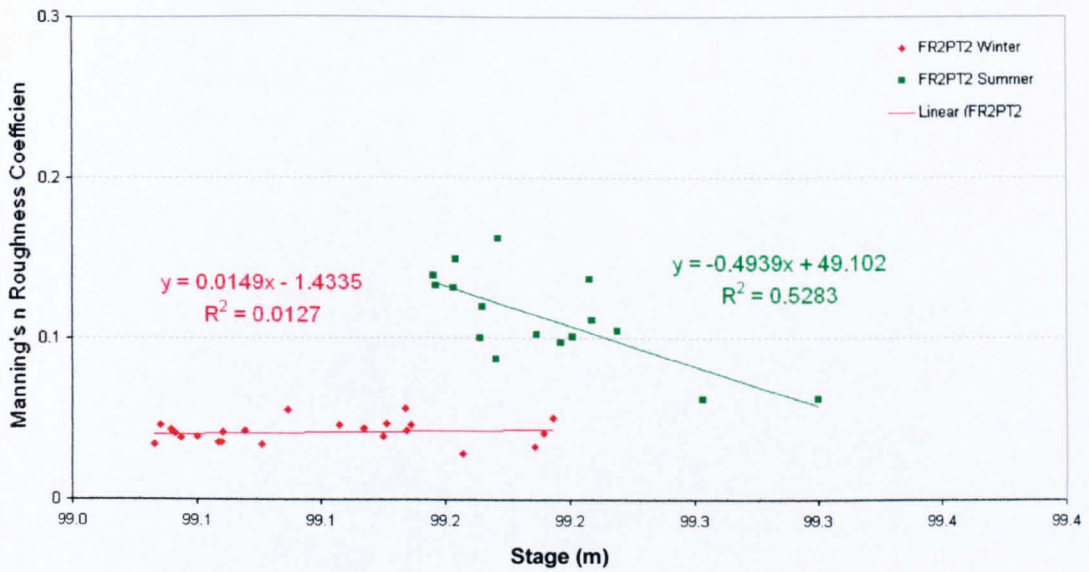


Figure 4.16 Stage/Mannings n at Frampton Estate, September 2003 – January 2005

value only refers to the average plant volume for the entire length of the River Frome and effects may be more substantial at highly vegetated sites such as Frampton. There is, however, no evidence to suggest that displacement will have a greater influence at higher discharge. Sediment accumulation beneath vegetation beds has been observed to be substantial in many chalk rivers (Sand-Jensen, 1998; Cotton *et al.* 2006 and Wharton *et al.* 2006) and this may have some effect on river stage. The effects of sedimentation are considered in more detail in section 4.7.3.

Another possible answer to the ‘discrepancy’ between stage and roughness trends lies in the interconnected nature of stage, roughness and velocity. The discrepancy only exists if changes in stage are viewed as a consequence of roughness, whereas roughness values can equally be viewed as a consequence of stage changes (Dingman, 1984). For example, the following scenario may be plausible: increased roughness introduced by plant growth will lead to a decrease in velocity, which in turn increases stage; but in doing so, this increased stage will bring about a decrease in roughness which will then increase velocity. Such tightly interwoven relationships may not be separable, especially when factors are considered only in pairs. It is important to remember that the ‘bulk-flow’ roughness coefficients can only assist in providing an overall appreciation of the seasonal changes in reach-scale roughness and the results are not fully interpretable.

4.7 Mean cross-sectional velocity

Figures 4.17a and 4.17b detail velocity/stage plots for the Crockways and Frampton field sites and Figures 4.18a and 4.18b show velocity/discharge plots. Average

velocity values for each discharge measurement were obtained by dividing discharge by the cross-sectional area and were evaluated with respect to both discharge and stage. As with the stage/discharge plots, the discharge/velocity and stage/velocity observations were subdivided into winter and summer groupings. The stage/velocity data suggest moderately strong regression relationships: Crockways summer R^2 0.65 ($P < 0.00$) Crockways winter R^2 0.66 ($P < 0.00$); Frampton summer R^2 0.56 ($P < 0.00$) and Frampton winter R^2 0.75 ($P < 0.00$), while the discharge/ velocity data necessarily displays very strong regression relationships: Crockways summer R^2 0.95 ($P < 0.00$) Crockways winter R^2 0.86 ($P < 0.00$); Frampton summer R^2 0.95 ($P < 0.00$) and Frampton winter R^2 0.92 ($P < 0.00$).

The patterns observed in the stage/discharge data are almost perfectly reversed in the stage/velocity and discharge/velocity plots with summer values plotting below winter values (see Section 4.6). The stage/velocity plots appear to show that mean velocities are greatly reduced in summer at Frampton, but that, again, the difference is much less apparent at Crockways. If the regression lines are regarded as representing the average velocity at a given stage height, then the average velocity reduction at Crockways (in summer relative to winter and at equivalent stage) ranges from an estimate of 0.06 m s^{-1} to 0.02 m s^{-1} , within the range of the available data (see Figure 4.17a).

At Frampton results are more manifest: average velocity reductions, from winter to summer, range from 0.20 m s^{-1} to 0.21 m s^{-1} at equivalent stage values within the range of available data (Figure 4.17b). If the winter regression line is tentatively extended, to cover the same stage range as the summer data, then the difference in

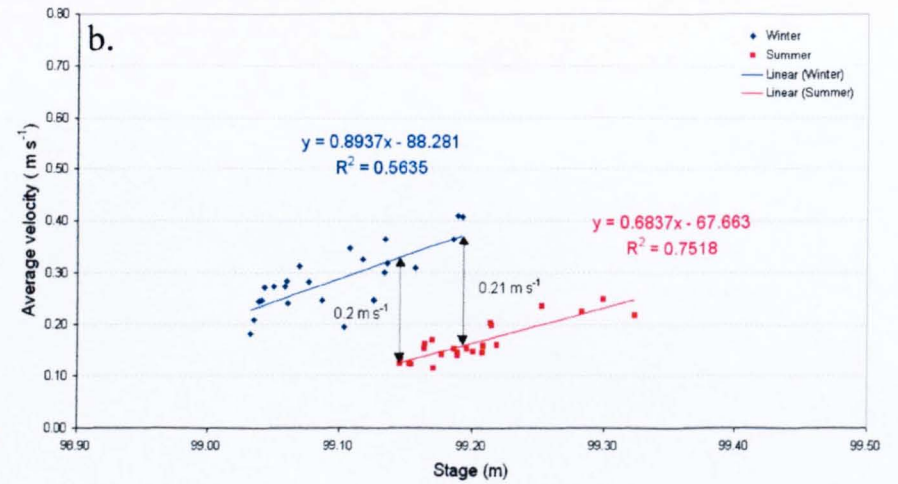
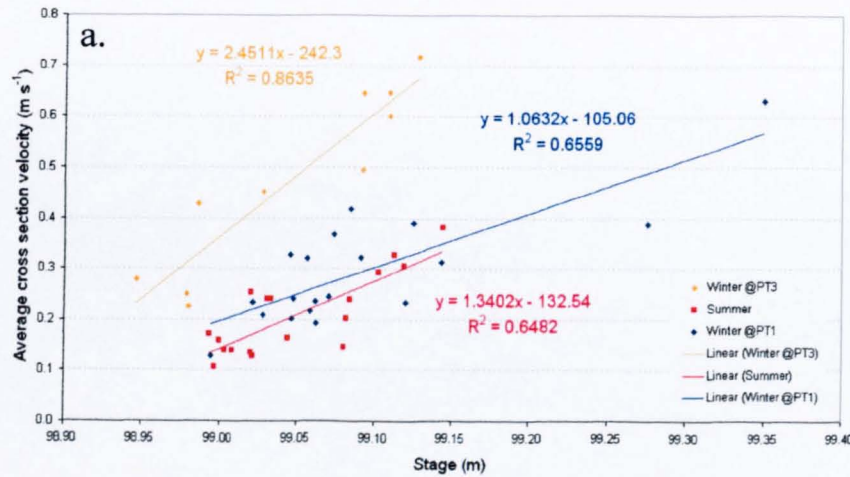


Figure 4.17 Stage/velocity plots for: a) the Crockways; and b) Frampton field sites, October 2003 – July 2005.

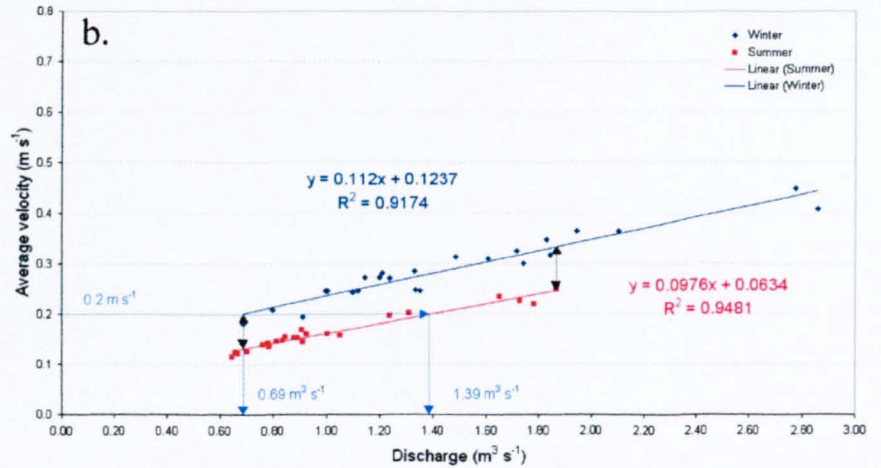
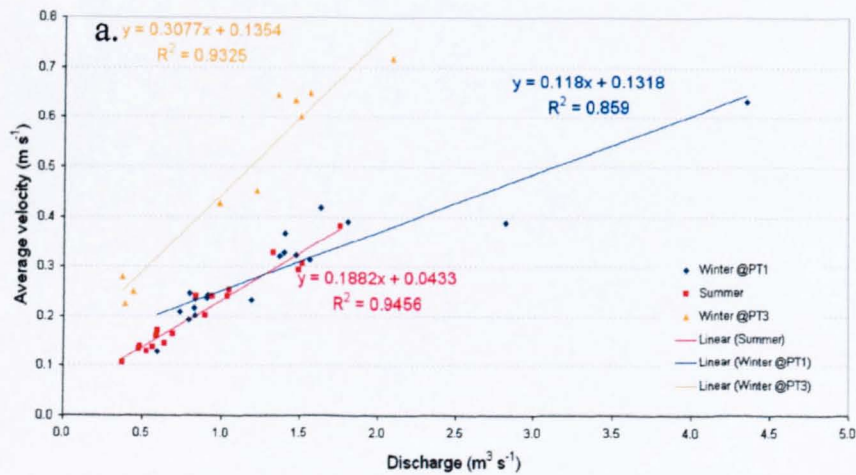


Figure 4.18 Discharge/velocity plots for: a) the Crockways; and b) Frampton field sites, October 2003 – July 2005

summer/winter velocities increases to 0.24 m s^{-1} (at equivalent stages of 99.32 m). Alternatively, if summer and winter velocities are examined at a single stage value of 99.16 m, the lowest average stage in summer (35 % of bankfull stage), then this corresponds to an average velocity of 0.13 m s^{-1} in winter and 0.33 m s^{-1} in summer, a difference of 260 %. There is again a difference in slope evident between the winter and summer regression lines.

At both Frampton and Crockways the slope of the summer regression lines are lower than those of the winter data, indicating that velocity does not increase as rapidly in summer with rising stage height as it does in winter or, put another way, that vegetation-induced velocity reduction is greatest at higher discharges. These velocity/stage results are in accordance with the stage/discharge data, where summer/winter stage differences were shown to be greatest at high discharges. The discharge/velocity plots (Figure 4.18a and Figure 4.18b) show similar trends to those displayed in the stage/velocity data and this provides an independent check on the automated stage data. As in the stage/velocity plots there is a reduction in average summer velocity relative to discharge, compared to the winter discharge/velocity relationship, and a noticeable difference in this effect between Crockways and Frampton. At Frampton a velocity of 0.2 m s^{-1} corresponds to a low discharge of $0.69 \text{ m}^3 \text{ s}^{-1}$ in winter (6.3 % of winter bankfull discharge), compared to a much higher discharge of $1.39 \text{ m}^3 \text{ s}^{-1}$ (12.7 % of bankfull winter discharge), a discharge difference of just over 200 %. One evident difference between the stage/velocity and discharge/velocity plots is the reduced scatter and higher R^2 values in the discharge/velocity data relative to the stage/velocity data. This might plausibly suggest that, as with roughness in Section 4.6, velocity reduction does not fully

explain changes in stage, and that other factors, for example scour and fill of the channel bed, may have an influence on stage levels.

4.8 Channel dimensions: width; depth and cross-sectional area

The final parameters from the suite of variables to be considered are those that describe physical channel dimensions: width; depth and cross-sectional area (Figure 5.19 to Figure 5.24). The time scales considered in the present study are too short to include major changes in the physical shape of the cross-sections, especially in a low energy chalk river context, and the changing dimensions considered at Crockways and at Frampton might be better described as changes in ‘water dimensions’, and not channel dimensions, as they describe the cross-sectional area of the channel occupied by water. Discharge/cross-sectional area and stage/cross-sectional area plots for Crockways and Frampton are presented in Figures 4.19 and 4.22. As expected, based on the preceding analysis, a change in the cross-sectional area occupied for a given discharge with season is highly evident at Frampton but less discernible at Crockways. The following sections explore the changing cross-sectional area in two dimensions: width and depth.

4.8.1 River width

There is only a subtle change in water width with changes in discharge at the Crockways field site and no discernible change at the Frampton field site (Figure 4.20 and Figure 4.23). This is not unexpected as the increase in channel width with

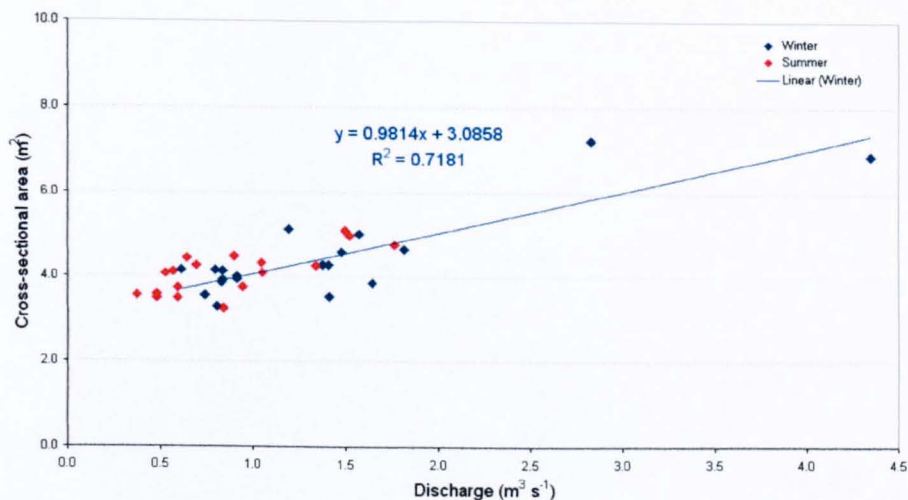


Figure 4.19 Comparison of cross-sectional area of the river and discharge at Crockways September 2003 – July 2006.

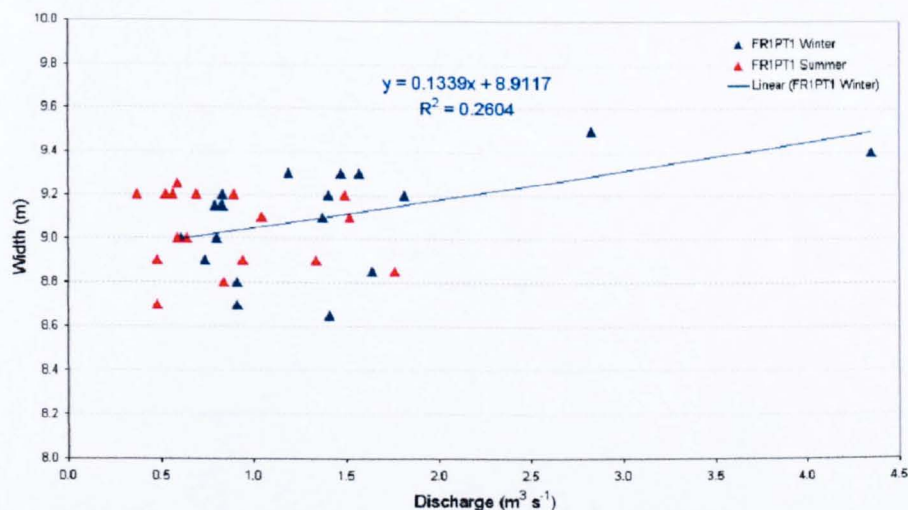


Figure 4.20 Comparison of river width and discharge at Crockways September 2003 – July 2006.

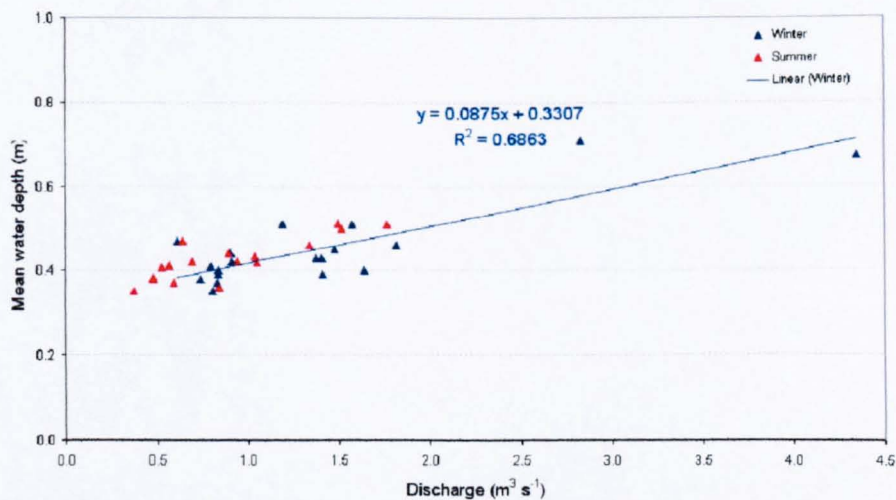


Figure 4.21 Comparison of water depth and discharge at Crockways September 2003 – July 2006.

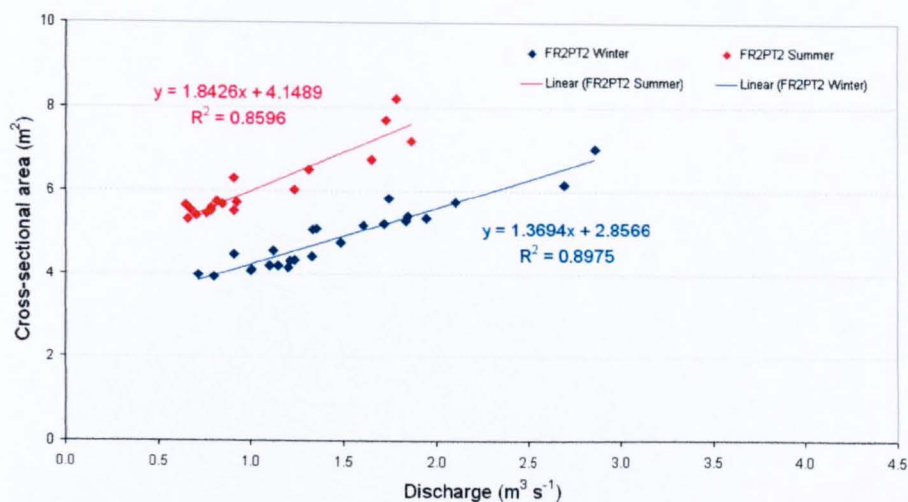


Figure 4.22 Comparison of cross-sectional area and discharge at Frampton September 2003 – July 2006

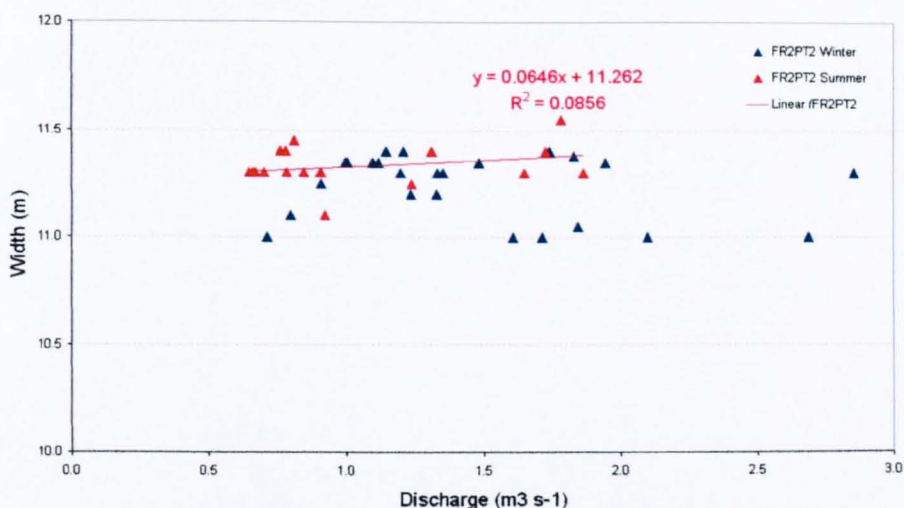


Figure 4.23 Comparison of water width and discharge at Frampton September 2003 – July 2006

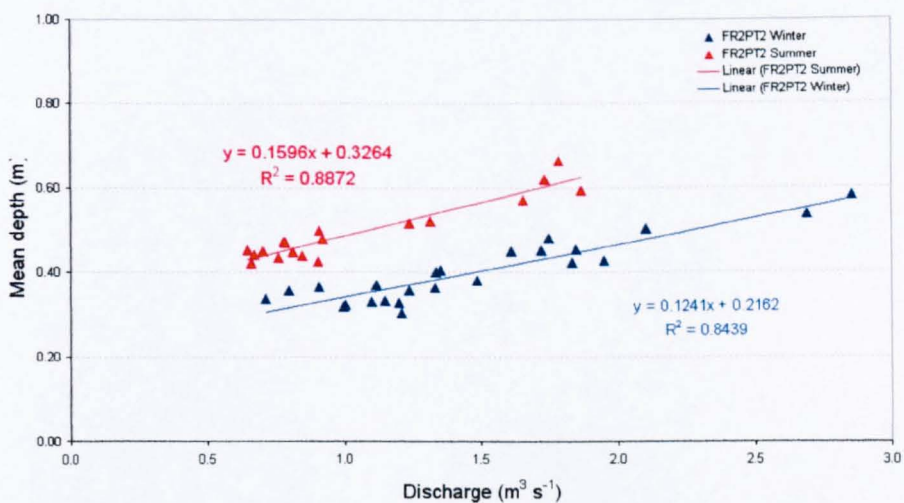


Figure 4.24 Comparison of water depth and discharge at Frampton September 2003 – July 2006

increasing discharge is generally found to be slight in most rivers (Dingman, 1984, Knighton, 1998). The Frampton data appears to be confined within a narrow range of water width, varying from 11.0 m to 11.5 m with no discernible pattern. This is consistent with the shape of the chosen cross-section at Frampton: the right bank is vertical and, sometimes slightly undercut in places, to a height of more than 1 metre, while the left bank is lower but is very steeply sloping. There is little scope for adjustment in water width at this cross-section and some of the variations in water width may reflect errors in measurement rather than any physical change: measurements of river width at the field sites were generally only accurate to the nearest 0.1 m. The Crockways cross-section also has vertical banks, but only to a height of ~ 0.8 m: after this point, the banks on both sides flatten out to a relatively shallow slope and this may be reflected in the discharge/width plot. Width values show no upward trend with increasing discharge until $2.0 \text{ m}^3 \text{ s}^{-1}$, after which two isolated points appear show a more marked increase in width with discharge. Overall, neither the Crockways or Frampton sites show any appreciable difference between the winter and summer width/discharge or width/stage relationships.

4.8.2 Water depth

Figures 4.21 and 4.24 show water depth and discharge plots in relation to average depth at Crockways and at Frampton. Average depth was chosen so as to minimise the influence of localised deposition and erosion on the depth measurements, which may affect measurements if taken in any one place. The depth/discharge plots are very similar to those found using the stage/discharge data (Figure 4.8 and 4.9) and provide an additional independent check on the automated PT stage data. The same

patterns are evident in the depth/discharge plots as found in the stage/discharge data: Frampton exhibits a major seasonal change in the depth/discharge relationship between summer and winter while Crockways experiences only a minor seasonal change in water depth. Unlike stage, however, water depth changes should only reflect the influence of velocity reduction and should not be affected by sediment changes: stage measures water surface elevation with reference to an arbitrary datum and includes sediment influence, while mean water depth is measured using the top level of the sediment in each survey as the reference datum. This difference between stage and discharge can be exploited to isolate the influence of sedimentation from that of velocity reduction. It will not be possible however to separate the roughness effects imposed by an uneven bed surface from the increased roughness afforded by the plants themselves.

At Frampton, both the slope of the regression lines (summer R^2 0.89; $P < 0.00$ and winter R^2 0.84; $P < 0.00$) and the absolute difference in water depth between winter and summer are surprisingly similar for depth/discharge as with stage/discharge. This suggests that the influence of vegetation on water depth at changing discharge may be very similar to the effect on stage; indicating that vegetation effects on water depth are greatest at higher discharges and also suggests that sedimentation changes, if they do occur, are virtually unaffected by discharge. The difference in water depth between summer and winter, at each end of the available data set, is 0.12 m – 0.18 m and this would seem to suggest that the summer influence of vegetation on water depth and water stage is virtually identical. On this basis, seasonal changes in mean bed elevation may be described as negligible, occurring at levels too small to be measured by this test. However, due to the differing methods of their measurement,

water depth and river stage are not entirely comparable. River stage is measured at one point in the channel cross-section, whereas mean water depth is the average of several depth measurements taken at different points within the cross-section. In addition, water depth is not as precisely measured as stage level; the PTs are capable of recording stage changes to the nearest 0.001 mV (allowing 0.01 m accuracy), whereas the depth measurements are only accurate to the nearest 0.1 m. The differences in measurement methodology and measurement error between the two data sets may have contrived to conceal differences between stage and depth.

Differences in measurement methods may have an influence when stage and water depth are compared remotely through stage/discharge and water depth/discharge, but this is no longer problematic if the stage and water depth variables are directly compared. Figure 4.25 and Figure 4.26 show depth/stage comparisons at Crockways and at Frampton. Both the Crockways and Frampton plots demonstrate that there are definite and repeated differences between the summer stage/water depth relationships (Crockways summer R^2 ; $P < 0.00$ and Frampton Summer R^2 ; $P < 0.00$) and the winter stage/water depth relationships (Crockways winter R^2 ; $P < 0.00$ and Frampton winter R^2 ; $P < 0.00$). At Frampton, stage is shown to be higher for a given water depth in summer than in winter. Given that methodological differences and measurement errors will apply to both the summer and winter data equally, then we must assume that there is a genuine summer/winter change in the relationship. This may reasonably be accounted for by changes in sedimentation patterns and in mean bed level changes through scour and fill, and support for sedimentation as a viable explanation may be obtained from comparison of the slopes of the regression lines. The slopes for summer and winter run almost parallel and there is, therefore, little

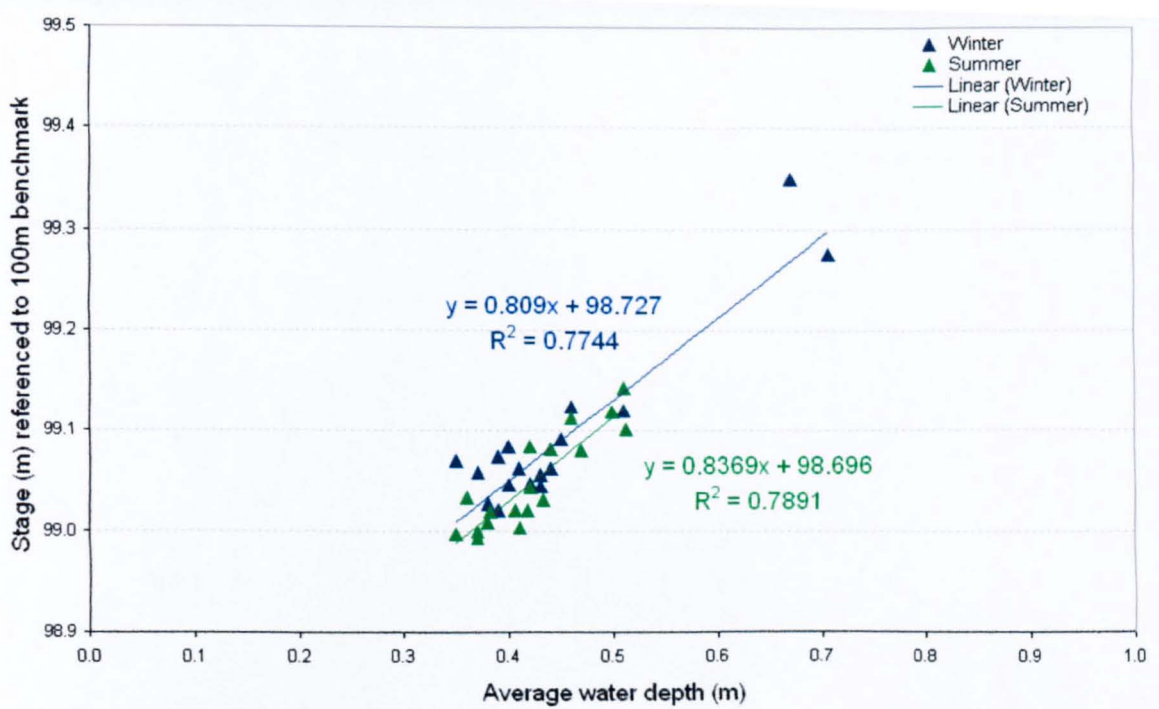


Figure 4.25 Comparison of river stage and water depth measurements at Crockways, September 2003 – July 2005.

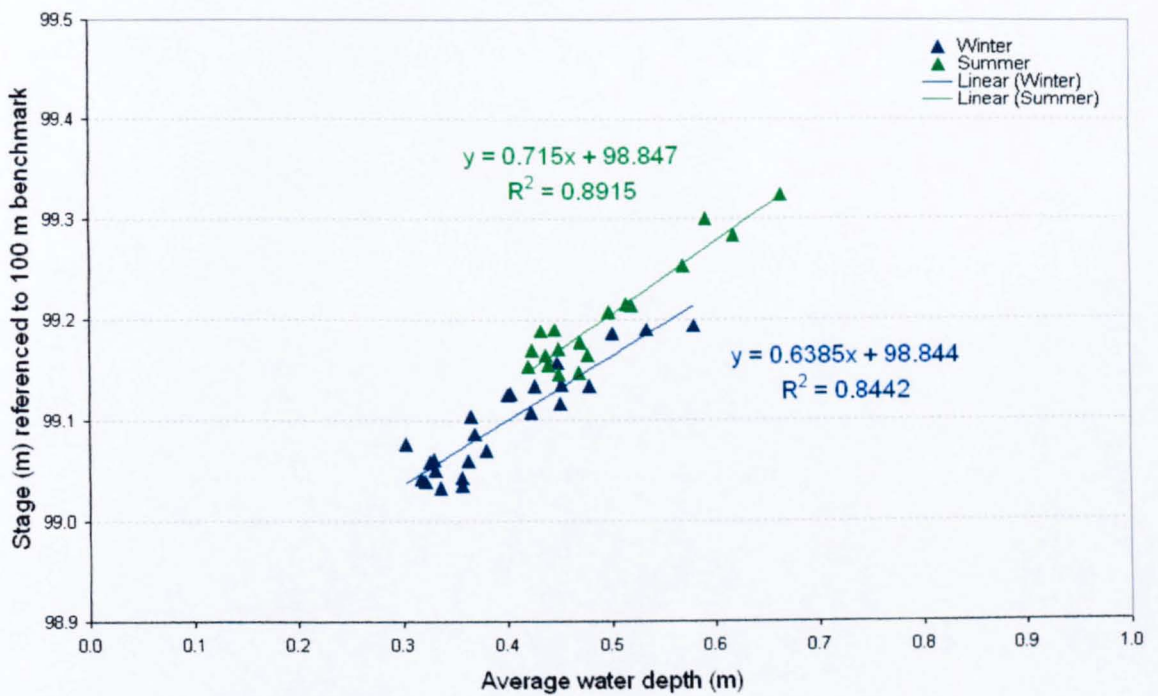


Figure 4.26 Comparison of river stage and water depth measurements at Frampton, November 2003 – July 2005.

change in the relative differences between summer and winter with changes in discharge, and it is reasonable to suspect that net sedimentation should be relatively unaffected by discharge changes: for example the slopes for stage/discharge and water depth/discharge were very similar meaning sediment effects most probably remain constant with discharge. If sedimentation is accepted as the major cause of change, then we can suggest that the increased summer elevation in bed level, relative to winter, at Frampton is in the order of 0.04 m and that sedimentation within vegetation beds, at Frampton is capable of increasing overall mean bed level. Personal observations at Frampton and at Crockways suggest that fine sediment accumulation within plant beds can reach significantly greater depths than this mean 0.04 m level and Cotton *et al.* (2006) report fine sediment depths as deep as 20 cm beneath *Ranunculus calcareus* plants on the River Frome. It would appear that erosion, or at least non-deposition, in un-vegetated flow channels is sufficient to partly, but not wholly, offset sedimentation within plant beds at the reach scale. This is indicated by the parallel regression lines and similar multipliers in the regression equations, observed between summer and winter in the stage and water depth plots.

The *relative* influence of bed level change on stage will actually decrease with increasing discharge, because stage itself does respond to discharge changes. The relative influence of bed level change is actually greatest at low flow (33 %), when the summer stage increase is lowest, and will be smallest at high flow, when summer stage increases are highest (24 %).

In the corresponding stage/depth plot for Crockways (Figure 4.25), there appears to be a small discernible *decrease* in mean bed level between the winter and summer

stage/water depth relationships of 0.01 m. This decrease is unexpected but is reasonable given the only minor summer reduction of velocity at this site. It may be that sediment is retained upstream in highly-vegetated reaches and this means less sediment is available in the un-vegetated reaches and/or that the water has more power to erode the bed in sparsely vegetated areas having deposited much of the suspended load upstream. If viewed as a proportion of total stage changes, then these sedimentation estimates reduce the water level increases through resistance at Frampton from 0.12 - 0.17 to 0.08 – 0.13 m, and increase the water level changes at Crockways, from 0.01 m to 0.02 to 0.02 – 0.03 m. The effects of velocity reduction and sedimentation at the two field sites are summarised in Figure 4.27.

4.9 Water surface slope

The impact of vegetation growth on roughness, velocity and river stage has received some attention in the past (Temple, 1991; Gurnell and Midgely, 1994; and Sellin and Keast, 1997) but as yet no research has explored the influence of vegetation on water surface slope. Past studies have generally focused on short-term changes in water surface slope which occur during changing discharge conditions, often related to changing energy gradients across long-term physical river features such as pools and riffles (e.g. Emery, 2003). This present study seeks to apply methods from previous morphological studies to the examination of seasonal changes in reach scale water surface slope.

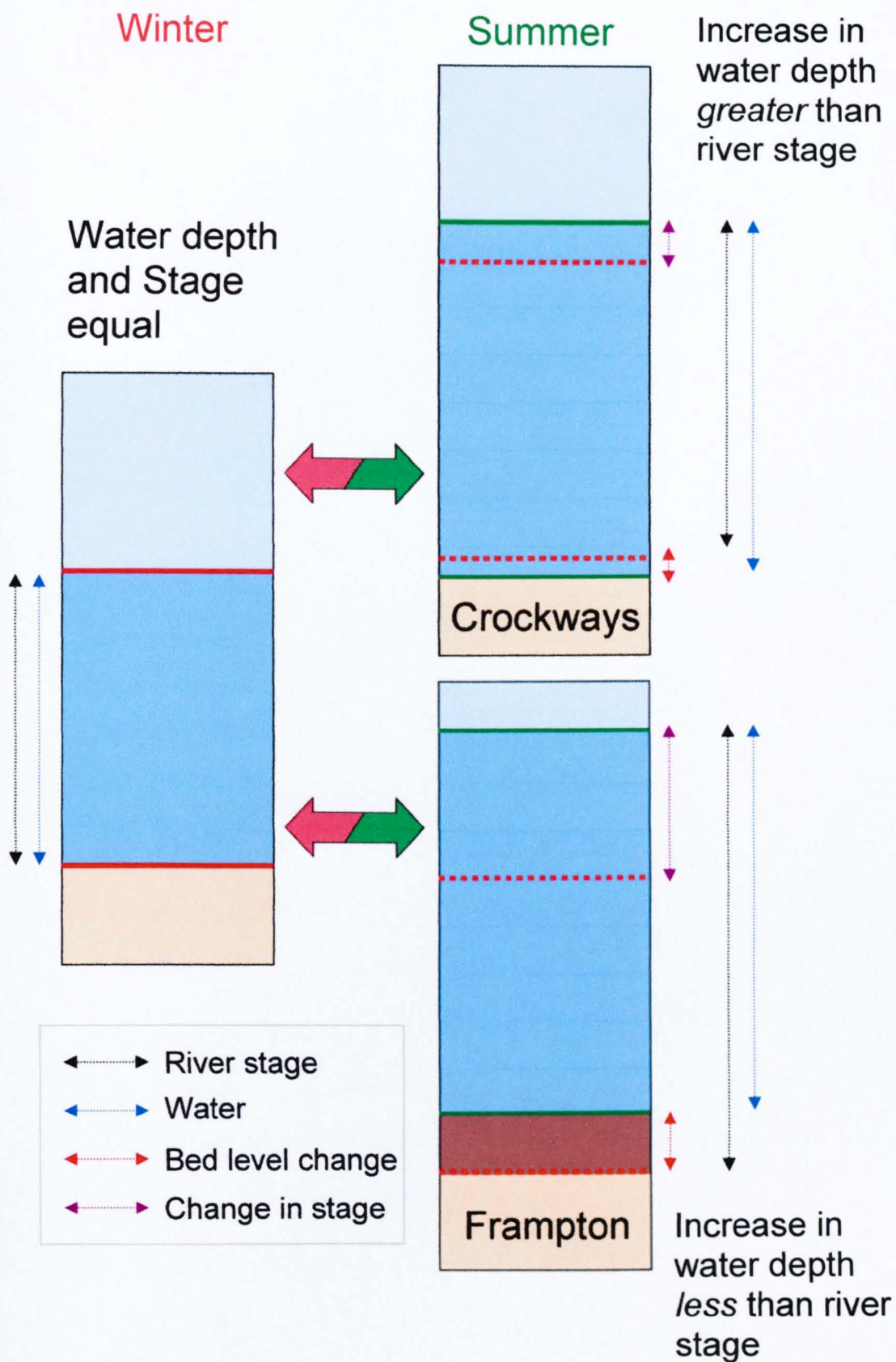


Figure 4.27 Summary diagram of water column changes at Crockways and Frampton based on the empirical data (diagram is not to scale).

4.9.1 Data transformations

The seasonal effect of vegetation on the stage record at individual PTs has been comprehensively studied above but as yet no analyses have been presented which link the three PTs at each site. Figure 4.7 has shown the calibrated PT records of stage averaged over one-hour intervals at Crockways and at Frampton and gave an indication as to the length of the data record and the seasonal rise and fall in stage as a consequence of the annual river regime. Figure 4.28 and Figure 4.29 show the stage record at each site after smoothing using 28-day running means. This removes small scale variation in the stage records imposed by storm events and allows the broad-scale seasonal relations between individual PT traces to be more clearly observed. It is evident from the graphs that the distances between the PT traces, representing the water surface slope, do not remain constant over the length of the PT record. Definite convergence and divergence of the PT traces can be observed and it appears that PT relationships oscillate according to season. This oscillating effect can be more easily examined if the *differences* between PTs are compared.

Figure 4.30 and 4.31 show the differences in PT stage readings at Crockways and at Frampton for two full seasonal cycles: from spring 2003 to spring 2005 and using hourly averaged data. The graphs have been partitioned into four ‘summer’ and ‘winter’ phases, based on the seasonal timings of vegetation effects evident in the stage/discharge relationships in the previous section (see Table 4.3). Each possible PT combination was compared within each site, and, in each instance, the upstream pressure transducer readings were subtracted from the downstream PT readings. Three ‘differenced’ traces were obtained: PT2-PT1; PT3-PT2 and PT3-PT1. Each of

these differenced values was then calibrated according to the distance between PTs using data obtained from total station surveys. This distance ‘averaging’ removes the influence of non-standardised distances between PTs, so that the PT differences may be directly compared. Distance averaging also means that the third PT combination (PT3-PT1) provides an average water surface slope for each reach to which the upper (PT2-PT1) and lower (PT3-PT2) sections of the reach may be compared. PT3-PT1 acts as a reference, or baseline, by which to judge the partitioned sub-reaches, and gives an indication as to how homogeneous water surface slope is within the reach. This is evident in Figures 4.30 and 4.31 where PT3-PT1 runs through the centre of the other PT traces at all times. The closer the sub-reach lines are to the reference line, the more homogeneous is the reach water surface slope.

4.9.2 Seasonal patterns in water surface slope

From consultation of Figure 4.30 and 4.31, it is apparent that both Crockways and Frampton are subject to repeated seasonal oscillations in the relationships between PTs and, hence, water surface slope. The character and timing of the cycles differs markedly between sites, and the sequence apparent at one site appears to be in exact negative phase with the sequence observed at the other site. During the summer season at Crockways, the water surface slope in the PT reach becomes increasingly homogeneous, whereas at Frampton the water surface slope is least homogeneous in summer. These states are then reversed in winter: Crockways becomes increasingly heterogeneous and Frampton becomes increasingly homogeneous. It is tempting to suggest that these changes represent differing vegetation influence on the morphological signal of each reach: at Crockways the complex morphological signal

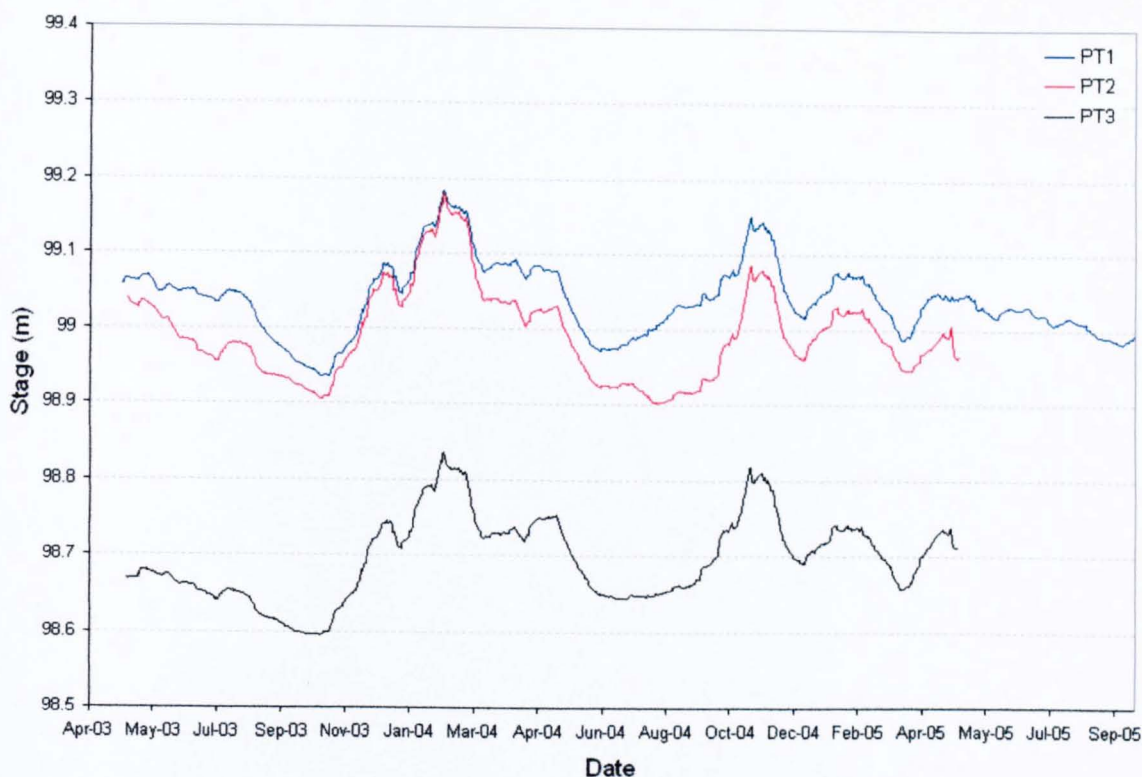


Figure 4.28 The stage record at Crockways after smoothing using 28-day running means to removes small scale variation in the stage records imposed by storm events.

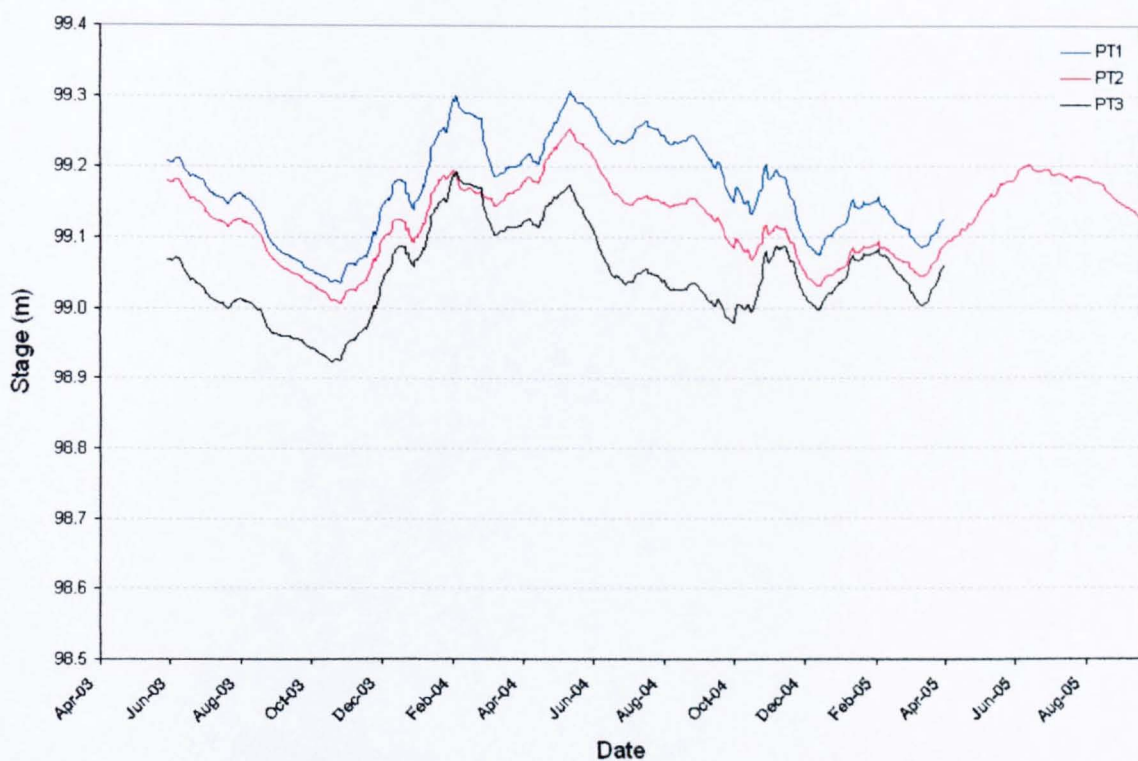


Figure 4.29 The stage record at Frampton after smoothing using 28-day running means to removes small scale variation in the stage records imposed by storm events.

Figure 4.30

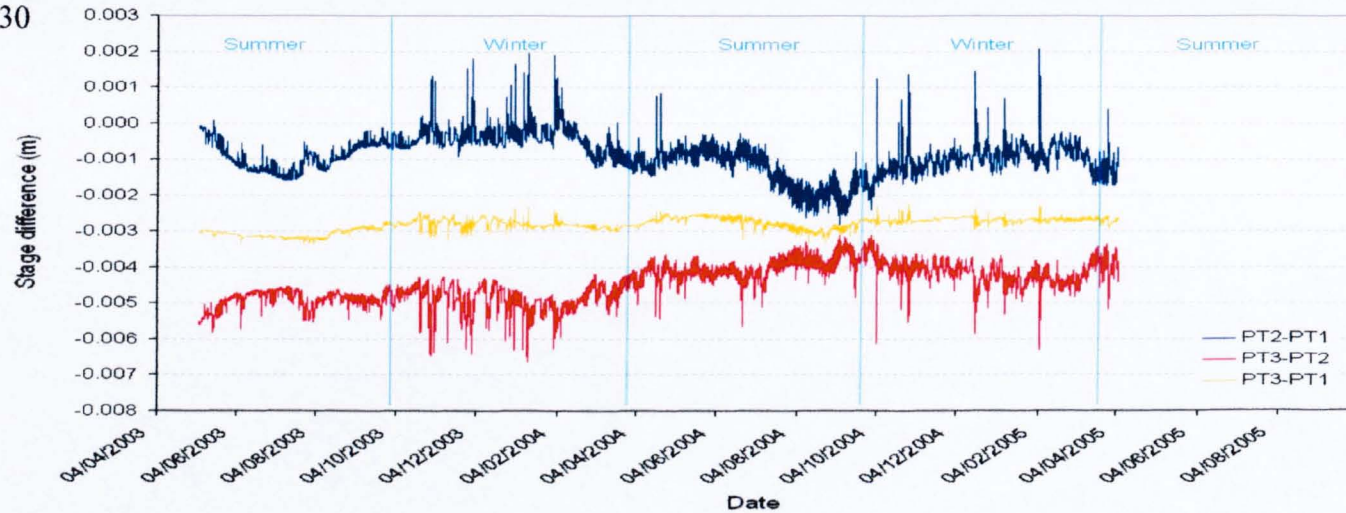


Figure 4.31

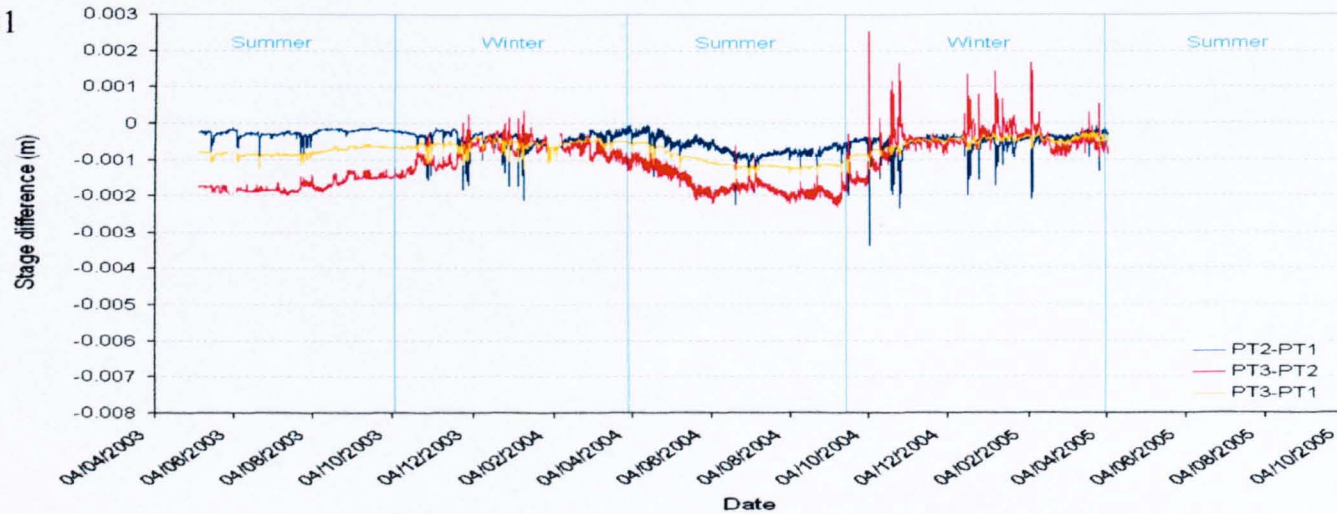


Figure 4.30 Differenced PT stage readings at Crockways and **Figure 4.31** Differenced PT stage readings at Frampton Estate.

may be modified by vegetation influence in summer to create a more homogeneous reach, while at Frampton the summer vegetation creates complexity and heterogeneity within a simple channel with a muted morphological signal. Gurnell *et al.* 2006 analysed a shorter section of the same dataset (February to October 2004), averaged using 28-day running means to minimise the impact of short-term flood events, and concluded that the cycles were related to temporal changes in flow resistance within the PT reaches induced by changes in vegetation biomass. However, due to the shorter data set utilised in their study, Gurnell *et al.* did not recognise that the seasonal cycles were negatively phased between sites.

The analysis of stage/discharge data has already shown that there is a difference in the magnitude and nature of vegetation influence on stage levels between sites. Frampton showed significant seasonal differences in the stage/discharge relationships between summer and winter, while at Crockways little discernible difference is evident between seasons. Based on the results of these earlier analyses, it could be theorised that the influence of vegetation on water surface slope occurs only at Frampton, and that the Crockways site might be regarded as a control reach by which to judge the vegetation effects evident at Frampton. However, if the differences between sites are to be thought of as a sign of different magnitudes of vegetation influence, with Crockways representing a site with stage levels virtually unaffected by vegetation growth, then why would a strong cyclical, and seasonal, trend of any form be evident at the Crockways site? One explanation might be that the trend is discharge related, with the cyclical water surface slope trend following similar seasonal patterns in the flow regime: high discharge in summer and low

discharge in winter. Emery (2003) describes similar oscillations or crossovers with changes in discharge at riffle-pool sites on the Afon Llwyd in Wales.

4.9.3 The influence of discharge on water surface slope

The theory of discharge influence can be assessed independently of season, but at smaller time scales, by considering changes in the water surface slope during flood events, either in winter or in summer. Figure 4.32 shows water surface slope changes in relation to a stage record of individual flood events at Crockways in January and February 2004 and visually illustrates that an increase in stage, and therefore discharge, corresponds to an increase in the heterogeneity within the PT reach. A similar comparison between stage and water surface slope at Frampton (Figure 4.33) reveals a more complicated relationship, whereby an increase in discharge initially leads to an increase in reach homogeneity but is interrupted by a ‘crossover’, after which reach homogeneity decreases with increasing discharge. These short-term variations can also be seen repeatedly in the traces in Figure 4.30 and 4.31, in both winter and summer, and they follow the same pattern as the longer term seasonal variations at each site. The longer-term 28-day averaging of the PT series employed by Gurnell *et al.* (2006) acted to remove shorter term variation and this meant that the link between water surface slope and changing discharge was not recognised.

To further explore the relationship between discharge and water surface slope, the stage difference at a number of known discharges were compared at each study site (see Figure 4.34 – 4.36) using both summer and winter data. Plots of discharge and

water surface slope at Crockways show a positive relationship for the sub-reach A (PT2 – PT1) and a negative relationship for the sub-reach B (PT3-PT2). Both regression relationships are relatively weak (R^2 of 0.18; $P < 0.07$ and R^2 0.11; $P < 0.037$, respectively), but some scatter may be explained by seasonal changes in the stage/discharge relationship (see Section 4.6) and by the predominance of low discharges in the dataset. The water surface slope is calculated as the downstream change in water surface elevation and thus values are generally given as negative numbers, and this means that positive and negative correlations must be interpreted differently: the positive relationship in sub-reach A means that water surface slope across this reach *decreases* as discharge increases, while the negative relationship in sub-reach B means that water surface slope *increases* as discharge increases. The opposing directions of the discharge/water surface slope relationships indicate that the water surface slope in the two reaches become increasing different as discharge increases.

It is possible that the local morphology of the sites has influenced the changing nature of the water surface slope across the Crockways reach. As noted previously, the PTs are located along a meander bend and across a pool-riffle sequence. This morphology may mean that the water surface slopes of the two sub-reaches differ greatly, especially as the central PT, which marks the division of the two reaches, is located at the inflexion point of the meander bend. This central PT neatly divides the upper, deeper, subsection from the lower, shallower, section. At low discharges this morphological division may be less important than at high flow and this may explain the changes in reach water surface slope in Figure 4.30. Several authors have studied

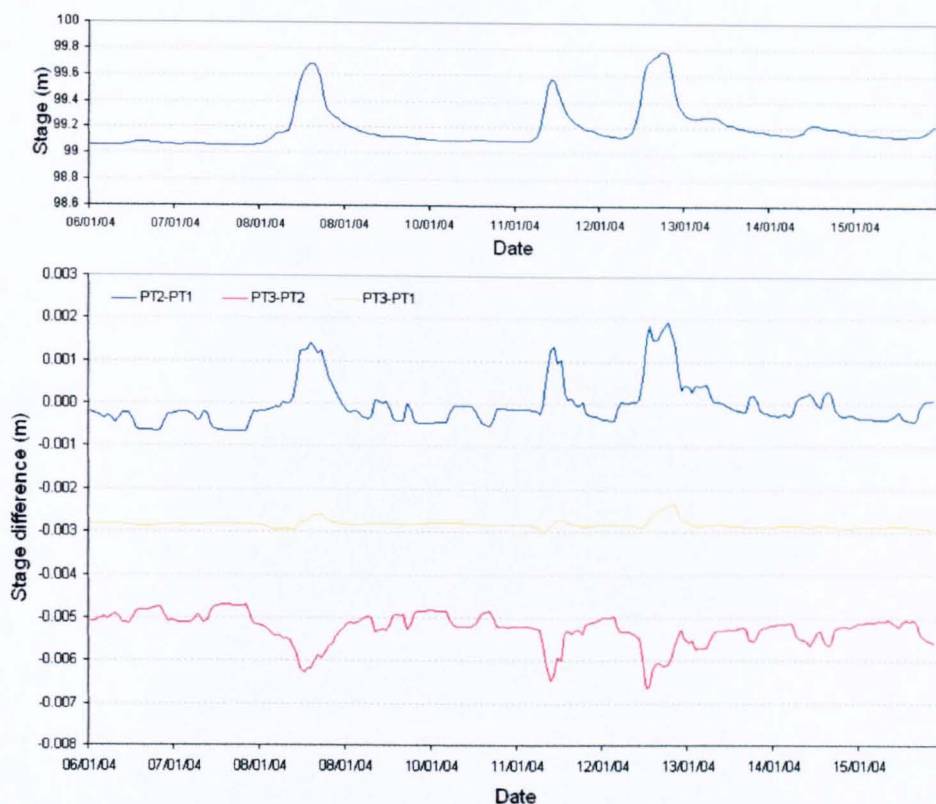


Figure 4.32 Change in water slope with discharge for consecutive flood events at Crockways, January 2004 (hourly data).

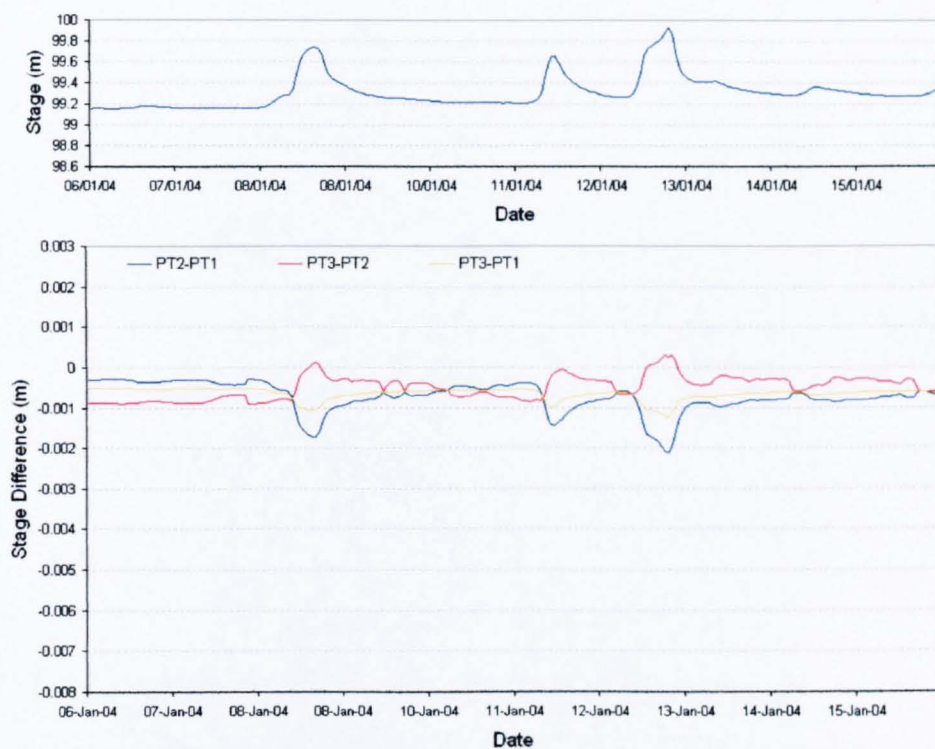


Figure 4.33 Change in water slope with discharge for consecutive flood events at Frampton, January 2004 (hourly data).

the changing water surface elevations above pool and riffle features and it is accepted that the position of maximum and minimum water surface slope changes across these bedforms as stage rises and falls (Lisle, 1979; Lisle, 1987; Jowett, 1993; and Emery, 2003). However, most pool and riffle research has suggested that morphological features were more influential on water surface slope at low flows because their effect was 'drowned out' at high flows and this is in opposition to the pattern observed at Crockways. It may be reasonable to suggest that a planform feature such as a meander bend may impart a markedly different response with increased discharge than a pool-riffle sequence on the channel bed. It may also be possible that the islands in sub-reach B have a greater effect at high flow and their influence may be transmitted upstream. Regardless of the physical explanation, it is evident that the seasonal cycles are predominantly discharge controlled and not induced by seasonal vegetation influence.

A similar comparison of discharge and water surface slope at Frampton (Figure 4.35) does not appear to show a coherent relationship between discharge and water surface slope for either sub-reach. This suggests that the seasonal cycles at this site may not be discharge related or that the relationship is complicated by other factors, which may include vegetation growth. In an attempt to isolate vegetation influence, the discharge and water surface slope data were separated into winter and summer plots (Figure 4.36). Figure 4.36a shows the winter data and suggests that a moderate negative relationship between discharge and water surface slope exists in sub-reach C (PT2-PT1), while a weak positive relationship may be discerned for the sub-reach D (PT3-PT2). Again, given the negative numbers used for water surface slope, this means that for sub-reach C the water surface slope increases with increasing

discharge while for sub-reach D the water surface slope decreases with increasing discharge. Initially, these opposing responses mean that the water surface slope, throughout the combined reach, becomes more homogeneous. However, a critical discharge is reached, between $1.5 - 2.0 \text{ m}^3 \text{ s}^{-1}$ (25.5 % - 34.1 % of winter bankfull discharge), after which the water surface slope becomes increasingly heterogeneous between sub-reaches. There are no major channel or planform features at the Frampton site and the winter relationship between water surface slope and discharge cannot be easily explained by morphology, but the actual changes experienced in winter are slight (0 – 0.002 m reduction in water surface elevation per 1 metre distance downstream), which may mean that even slight morphological variations are sufficient to have an effect.

The relationship between discharge and water surface slope in summer is shown in Figure 4.36b and presents a different pattern to that found in winter. Initially, the data for both sub-reaches suggests a positive trend, meaning water surface slope decreases with increasing discharge and the reach becomes increasingly homogeneous. However, the rate at which water surface slope changes with discharge is different for each sub-reach and the two sub reaches become increasingly similar up until a critical discharge of around $1.05 \text{ m}^3 \text{ s}^{-1}$ (9.6 % of winter bankfull discharge) when a cross-over occurs and the water surface slope in the overall reach becomes more heterogeneous. At discharges greater than $1.3 \text{ m}^3 \text{ s}^{-1}$ (12.2 % of winter bankfull discharge), the data suggests that water surface slope and discharge are *negatively* correlated for both sub-reaches, meaning that water surface slope increases as discharge increases. This change in the direction of the sub-reach

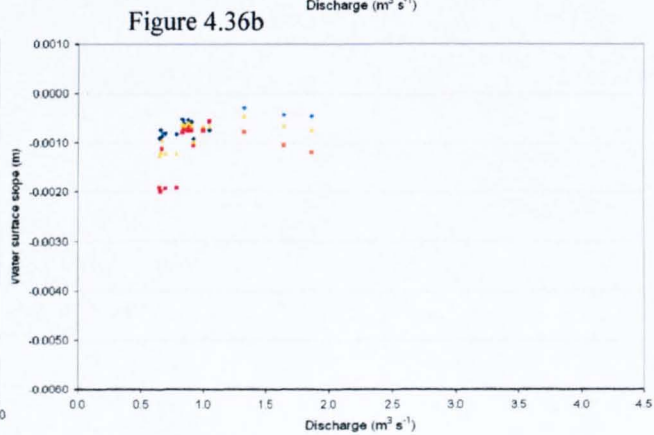
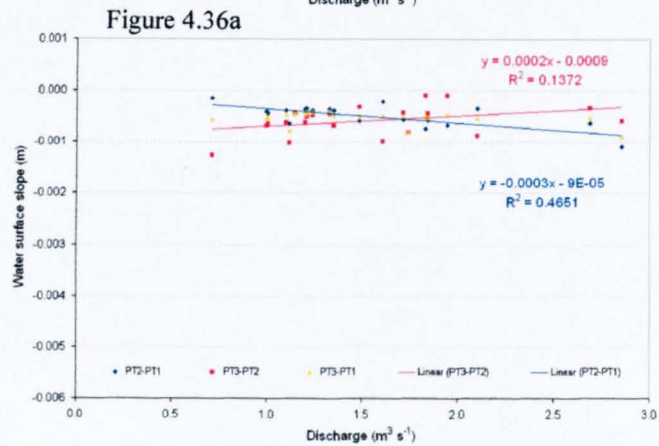
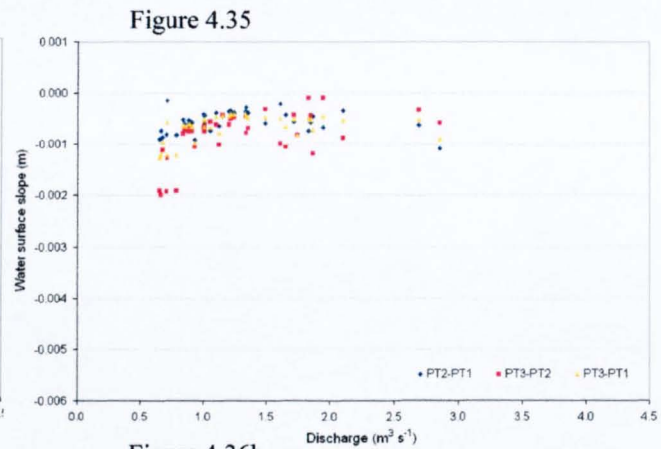
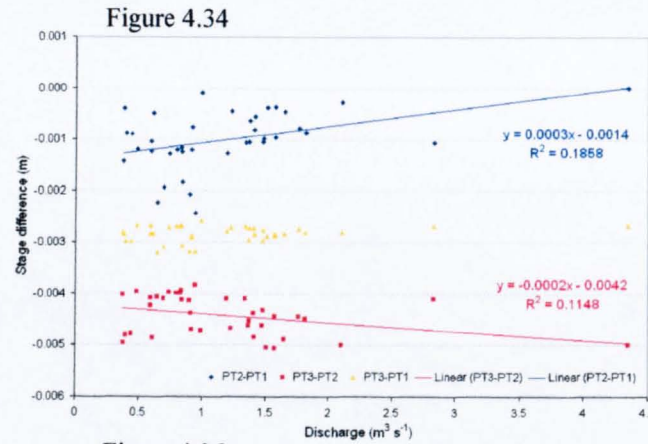


Figure 4.34 Crockways water surface slope/discharge winter and summer data combined.

Figure 4.35 Frampton WSS/discharge winter and summer data combined.

Figure 4.36 WSS/discharge: a) winter Frampton b) summer Frampton

relationships indicates a change from an initial phase where increasing discharge serves to reduce the water surface slope, perhaps indicative of an energy reduction and ponding of water flow behind vegetated sections, followed by a phase where water surface slope increases with increasing discharge, and which perhaps represents the bending and compaction of plants and a release of stored potential energy. The upstream sub-reach is more shaded and has less vegetation growth than the lower sub-reach, which is un-shaded, and this may explain the differences observed between sub-reaches.

It is clear from the presented data that water surface slope at both field sites is influenced by seasonal change. At Crockways these seasonal changes are thought to be discharge dominated, with changes in water surface slope following the predictable annual regime of a chalk river: high discharge in winter causing increased heterogeneity in the reach and low discharges in summer promoting homogeneity in the reach. The discharge influence is thought to be a consequence of the complicated morphology in this semi-natural reach. It is possible that seasonal vegetation effects at Crockways may exacerbate, or even reduce, the water surface slope changes observed in summer and winter but it appears likely that any effects are overshadowed at Crockways by dominant discharge and morphology effects. At Frampton, a relationship between discharge and water surface slope is also thought to exist (Figure 4.36), but when examined in more detail this relationship differs between summer and winter. All explanations given for the relationship between water surface slope and discharge in such a uniform reach are tentative, but it is obvious that a difference exists between summer and winter at Frampton, and that the most likely factor in this change is vegetation influence. The changing water

surface slopes at Frampton also imply that the change in stage is variable spatially throughout the reach and that these analyses based at two cross-sections may reflect only the local conditions at, or close to, the actual cross-section. The spatial variability of vegetation influence is explored in more detail in Chapter 5 using grid-based data.

4.10 Discussion

The analysis in this chapter has presented two river reaches which are located within a 2.5 km stretch on the same river, but which display marked differences in morphology and in the seasonal influence of vegetation. The results are sufficient to define quantitatively the effect of macrophytic vegetation on river stage, hydraulic roughness, mean water velocity, water depth, water surface slope and mean bed level and these reveal marked contrasts between the two sites. Vegetation was shown to have a significant and relatively simple effect on hydraulic parameters at one site, but a less obvious and possibly more complex effect at another site, in response to differences in channel morphology, vegetation abundance and vegetation type. Vegetation influence was also shown to vary with discharge, but not in the manner prescribed by the literature: net summer vegetation influence on stage was shown to be greatest at high discharges which may have important implications for summer groundwater levels and summer flooding. It may not be possible to directly relate the traditional n -VR diagrams, as presented by Chow (1959) and Watson (1987), to changes in stage, particularly at sites where emergent vegetation is abundant.

At Crockways, it is uncertain whether a true increase in stage occurred between winter and summer: the winter and summer data sets overlap and the regression relationships suggest that there is no difference between winter and summer stage at low flow and only a 0.02 m difference at high flow. This lack of seasonal change is repeated in the summer and winter plots of hydraulic roughness, water velocity and water depth, and it appears that, though there has been a change in vegetation biomass between winter and summer, there has been no change in reach-scale hydraulics. The increase in hydraulic roughness and reduction in velocity inside vegetation beds has not been sufficient to reduce mean velocity and it is likely that increases in velocity outside plant beds has compensated for reductions within. This suggests that a critical biomass does exist, which must be attained before vegetation effects become evident, and which was not reached at Crockways due to the complex morphology and variable depth at this site.

Sedimentation and bed level *is* believed to have changed between seasons at Crockways, with a mean *reduction* in bed level from winter to summer of 0.01 m. This decrease is thought to be a consequence of higher vegetation biomass upstream of the PT reach: suspended sediment will be deposited within the high biomass areas upstream, meaning less is available for deposition in the less vegetated Crockways reach, and the resulting low sediment load will mean that river flows will have a higher capacity to erode in the Crockways reach. This reduction in bed level also suggests that a critical vegetation biomass was not attained at Crockways, as increases in fine sediment depth observed within the plant beds must have been offset by erosion outside. This seasonal bed level change means that the influence of vegetation on water depth was underestimated by the stage/discharge analysis, and

suggests that *water depth* increased by 0.01m – 0.03 m from winter to summer within the range of the available discharge data.

The Frampton data, in contrast, show a marked increase in river stage from winter to summer of 0.12 – 0.17 m. These increases in stage are mirrored by an increase in water depth (0.12 – 0.18 m), an increase in average roughness (Manning's n average 0.12 in summer, 0.04 in winter) and a major decrease in velocity ($0.20 - 0.21 \text{ m s}^{-1}$) from winter to summer. This marked influence on hydraulic parameters suggests that a critical biomass has been attained at Frampton. This critical biomass marks the point when compensatory increases in velocity outside of plant beds are no longer sufficient to offset the velocity reduction within plant beds, causing a reduction in mean velocity, and this leads to an increase in river stage. The exact nature of this critical biomass and the underlying spatial velocity patterns induced by the vegetation are investigated in more detail in Chapter 5. At Frampton, the switch between the winter and summer stage/discharge regimes appears to have been quite rapid with only a short transition period (Table 4.3) and this also suggests that a critical biomass was reached, after which vegetation effects were evident. Mean bed level at Frampton was observed to *increase* by 0.04 m between winter and summer as a consequence of trapping and storage within plant beds. This is a further indicator that a critical biomass was attained at Frampton. The seasonal change in mean bed level means that the stage/discharge analysis has overestimated the effect of roughness on river stage and this influence may be isolated as contributing only 0.08 – 0.13 m to stage increases.

The site differences in the present study and the wide-ranging estimates in the published results (Table 4.2) highlight the complicated nature of vegetation influence on river stage and other reach-scale hydraulic parameters. The present analysis has revealed that the stage increases at a site will be determined by the nature of the critical biomass limit at a site and whether or not this critical biomass is attained. If the critical biomass is attained, then the magnitude of the stage increase will be controlled by several different physical factors: the vegetation abundance at that site, which itself is controlled by channel morphology and riparian shading; vegetation type; channel shape (width/depth ratio and planform characteristics); seasonal changes in sedimentation; and the range of discharges over which measurements are taken. These diverse contributing factors mean that it is essential that estimates of winter/summer stage increases be couched within their site context.

The analyses in the present study provide detailed information about the influence of vegetation on river stage, and contributing hydraulic parameters, and provide contextual information for each site. Vegetation abundance and riparian shading at the study sites have been quantified using RHS data, and have been examined in relation to other chalk river sites and river sites of other geology (Chapter 3, Section 3.5 and Section 3.6), vegetation type has been comprehensively described (Chapter 2, Section 2.6), and a description of channel shape at the study sites, in relation to the UK and chalk average has also been given (Chapter 3, Section 3.4). In addition, the analysis in this chapter has provided quantitative evidence and estimates of sedimentation effects on stage and the discharge range covered by the results has also been listed. These additional descriptors allow the river stage estimates to be utilised

at other sites on the River Frome and on other chalk rivers and permit greater certainty when used in management applications.

The site differences evident in this analysis, and the complicated factors which contribute to these differences, may appear problematic for traditional management approaches. The site differences suggest that a blanket management approach applied to large sections of the river will work at some sites but not at others and will not be necessary at others. This means that at some sites resources are wasted cutting vegetation where no cutting is needed, while at other sites management may fail to reduce biomass to below the critical level and will not control flooding. However, site differences could be viewed as advantageous: sites where critical biomass levels are not likely to be met and where no vegetation effects are likely can be left untouched, while management resources can be concentrated at sites where a critical biomass is likely to be achieved and where management is most needed. The estimates of stage increase in this study are given in their physical context and this allows the estimation of stage impacts on other rivers, e.g. if the width/depth ratio is lower at a proposed management site than at Frampton then stage increases are likely to be higher, if all other factors are equal. Sites which may require management could initially be differentiated by desk studies which examine the morphology of the reach and the degree of riparian shading, for example using the RHS database, and could be refined through field surveys of vegetation in selected reaches. In addition, the installation of a simple discharge cross-section station, fitted with a stage board, would allow inexpensive comparisons of stage height and discharge. The analysis in this chapter shows that stage increases from summer to winter, and with changing discharge, are consistent over two water years and this suggests that

the viability study need only be completed once, with periodic reviews of site changes.

5. REACH-SCALE PATTERNS IN VELOCITY AND FINE SEDIMENT

5.1 Chapter synopsis

This chapter aims to investigate the averaged seasonal trends identified in Chapter 4 and seeks to identify and quantify detailed patterns in vegetation cover, water depth, water velocity and sediment depth in the grid-scale data. The chapter also explores the results obtained through several different analysis techniques and examines the information and linked conclusions which emerge from each analysis. The grid-scale surveys were analysed in three linked stages of varying scale and complexity. The first analysis deals with each survey in totality and considers agglomerative summary measures of each variable over the survey reach. The second segregated analysis considers the surveys on the basis of: (i) each individual cross-section in selected data sets; and (ii) individual longitudinal transects, placed stream-wise through the grids, in selected data sets. Finally, the third analysis bridges these two earlier scales by considering all the observations as a whole, but in such a way as to consider each grid cell in relation to its neighbour and to develop a classification of individual grid cells or patches of cells. This third analysis was achieved using hierarchical agglomerative cluster analysis.

5.2 The grid-based method and the chosen grid reaches

Each grid-survey involved measurement of nine variables: river discharge; water velocity in three dimensions (x, y and z); water depth; sediment depth; total

vegetation cover; and vegetation type, recorded as either 'emergent' or 'submergent'. Measurements were conducted using a grid-based methodology, as outlined in Chapter 2, whereby a 20 m length of channel was divided into 1 m² grid cells. Measurements of water velocity, water depth, sediment type, and turbidity were taken as point measurements in the centre of each grid cell, whereas vegetation cover, vegetation type and sediment type were considered over an entire grid square. The two sites utilised for the grid-based measurements were the Crockways and Hydeclift plantation study sites. The third site, Frampton, was discounted from grid-scale analysis because of the very high in-channel vegetation abundance: the high vegetation cover made it an interesting location to examine reach scale hydraulic effects, as investigated in Chapter 3, but would have proved problematic during grid-based measurements. With a grid resolution of 1 m², distinctions between open and vegetated areas at Frampton would be impossible and a finer grid was considered prohibitively time-consuming. It was thought that the Hydeclift site would be more suitable for the grid-based work and would provide a more meaningful contrast to the Crockways site, while still helping inform analyses carried out at Crockways and Frampton in the previous chapter. Crockways and Hydeclift provide the contrast of: (i) an un-shaded and abundantly vegetated reach with a diversity of plant growth form; and (ii) a shaded, less abundantly vegetated, and less diverse reach. It is acknowledged that the use of the Hydeclift site for the grid-based survey will mean that the findings from Chapter 4 and Chapter 5 will not be directly comparable, but the principles governing vegetation influence should be transferable between sites. The general characters of both Crockways and Hydeclift, and the specific qualities of each grid-reach, have been comprehensively described in Chapter 2.

In total, 18 grid-surveys were completed: nine at Crockways, and nine at Hydeclift plantation, between January 2004 and January 2005. Each site survey generally took two days to complete; water velocity and depth were measured together on one day, while vegetation and sediment investigations were conducted on a separate day in the same week. The majority of the surveys are 20 m long and, on average, 9 m wide at Crockways and 10 m wide at Hydeclift, yielding around 180 and 200 observations at each site respectively. At Crockways, three of the nine surveys are incomplete; the Crockways reach is relatively deep, and under flood conditions in January 2004, April 2004 and January 2005 part of the reach was inaccessible and measurements were only possible for the first 10 metres of the grid. Hydeclift, in contrast, is a very shallow site and was accessible under high flow conditions and all surveys for Hydeclift are complete. The dates of each survey and the flow conditions in the channel during the surveys are detailed in Figure 5.1. Discharge at the sites on the day of the velocity and water depth surveys are shown in Figure 5.2.

5.3 Agglomerative analysis

Exploration of summary survey variables was achieved using three exploratory data analysis techniques: bivariate scatter plots, box and whisker plots and rose diagrams. These methods of analyses were preferred to surface maps of the data as they allowed more quantitative comparisons to be made and the data were considered as original measured data and not as interpolated data. Box and whisker plots were used to convey as much detail as possible about the structure and distribution of individual variables, while bivariate scatter plots considered variables in pairs. Not every variable was appropriate to include in box and whisker plots and only five

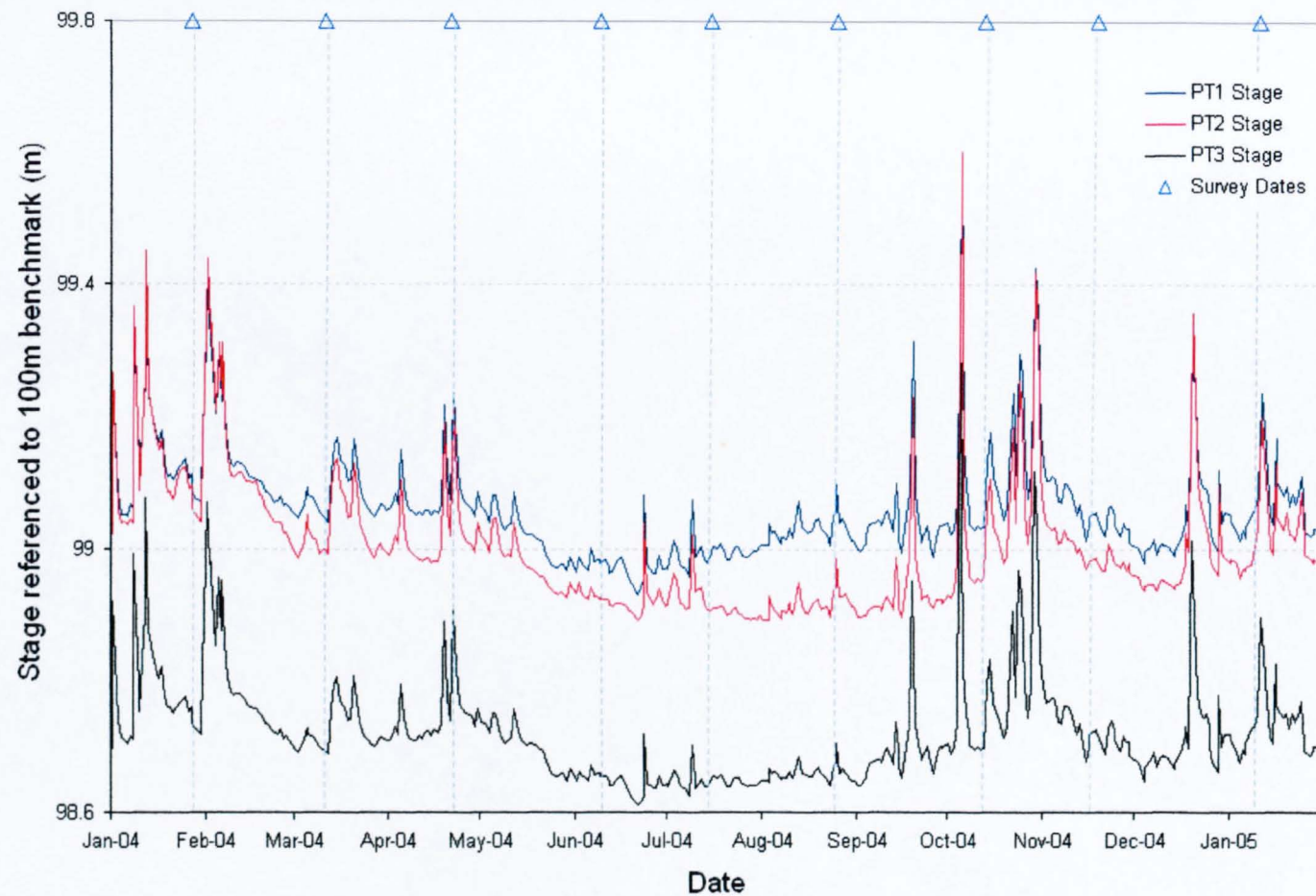


Figure 5.1 Grid survey dates January 2004 – January 2005 at the Crockways and Hydeclift plantation field sites, with reference to stage levels recorded at Crockways.



Figure 5.2 Discharge at the Crockways and Hydeclift sites for each of the grid-survey dates measured at cross-section 1 of each grid reach, January 2004 – January 2005. The change in discharge during the October Survey is evident in the difference between discharge measurements taken at the beginning and end of the survey.

parameters were considered: discharge; stream-wise velocity (V_x); re-scaled V_x (V_x was re-expressed on a scale of 0 -1); water depth; categorised vegetation cover; and fine sediment depth. Figures 5.3 (Crockways) and Figure 5.4 (Hydeclift) display the box and whisker plots, separated by variable. The plots are presented one above the other to aid comparison and each individual plot is ordered chronologically, from left to right, from January 2004 to January 2005. The main box shape of the plot represents the inter-quartile range, bounded by the upper and lower hinge (the 25th and 75th percentile respectively) and the whiskers extend to 1.5 times this inter-quartile range. The mean is represented by a small square symbol and the median by a straight horizontal line across the width of the box. Values greater than 1.5 times the inter-quartile range are represented by short horizontal strokes outside the main data body. The box-plots were produced using Origin Professional statistical software.

Rose diagrams, or circular histograms, were used to analyse cross-stream (V_y) and vertical (V_z) velocity (Figure 5.5 – Figure 5.8): these two variables were not appropriate for use in the box plots as negative and positive values of V_y and V_z would act to cancel each other out. The cross-stream and vertical velocity components were combined with the stream-wise velocity and converted into angles of deviation from 0°. If 0° is considered to constitute water flow parallel to the riverbanks (V_x) or to the channel bed (V_z) then any deviation from the baseline can be measured from 0 - 360° and displayed in a Rose diagram. Rose diagrams are similar to conventional histograms but permit easier interpretation of circular data. For example, a flow angle of 0° and 359° degrees are separated only by 1° and appear as such in the rose diagram but would be widely segregated in a conventional

histogram. The rose diagrams also provide a summary measure of the average flow direction (including 95% confidence limits) for the whole grid-reach, represented in the diagrams by a heavy black line. The variation of angles in each survey were explored to help to display the changing diversity of the velocity flow field throughout the year. The rose diagrams were produced using Oriana 2, a statistical package specifically designed for 'circular' data.

5.3.1 Agglomerate analysis: vegetation abundance

The use of categorised vegetation data has reduced the detail of the information available but strong general trends are still evident in the data at both sites. Both Crockways and Hydeclift exhibit a smooth seasonal trend, with a gradual increase in vegetation cover from a low in January/March 2004 through to a high in August and a return to a new low in January 2005. On average, winter vegetation cover was below 1-5% at both sites, while the average summer vegetation cover was just over 25 – 50% at Crockways and only 6 – 25 % at Hydeclift. Reach averaged values, especially using categorical data, are of limited value without also considering the spread of the data about the survey mean, which indicates the variability in vegetation cover between individual cells. The box-plots indicate that the inter-quartile ranges of the vegetation data are relatively similar at both sites in winter, but are much larger at Hydeclift than at Crockways in summer, indicating uneven vegetation cover at Hydeclift. Dawson (1976) suggested that peak biomass for *Ranunculus* plants, in British rivers, typically occurs in July, when the plants are in flower. However, *Ranunculus* was not the only macrophytic plant under consideration in this study and the vegetation measured in this study was not purely

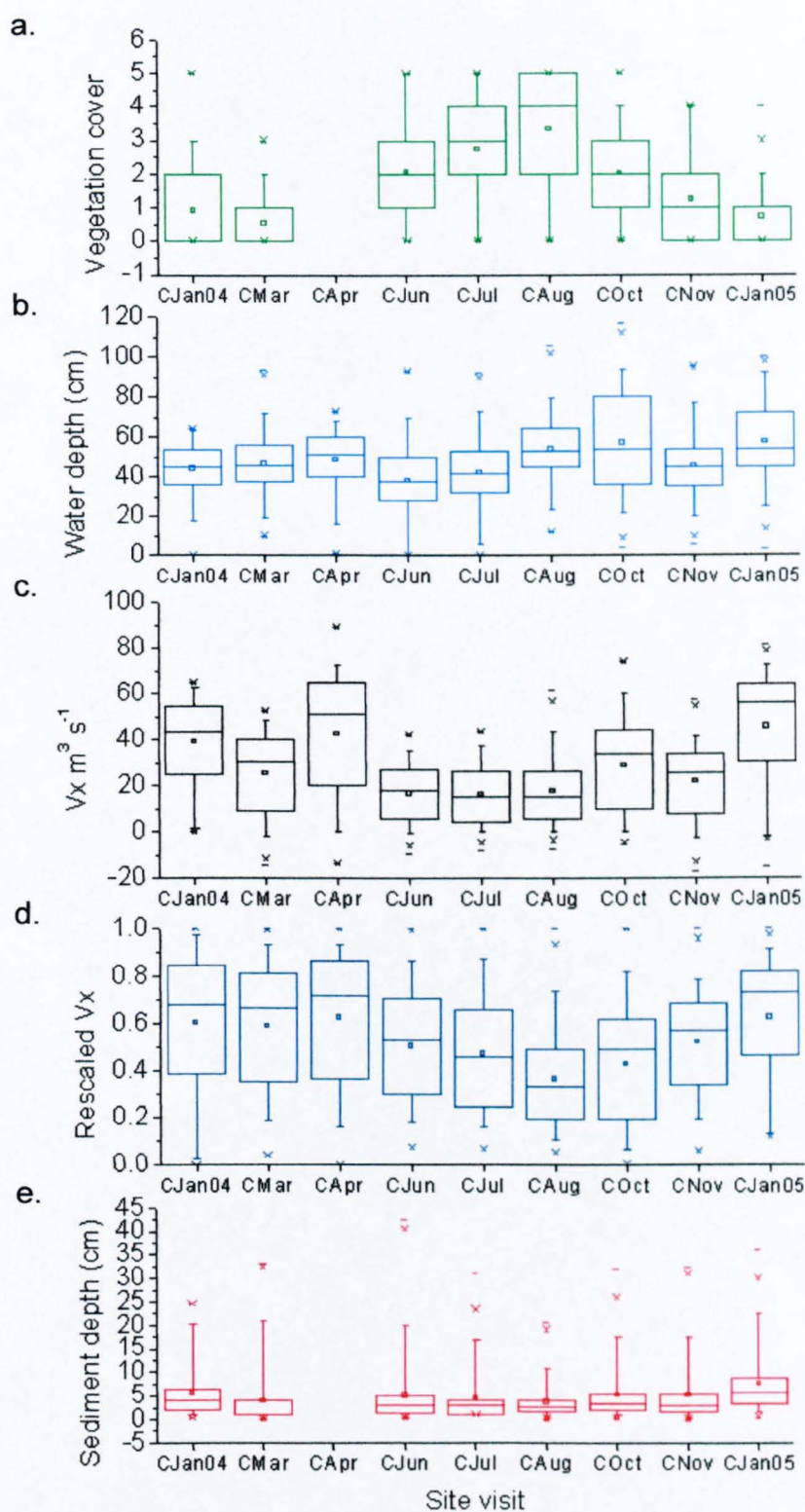


Figure 5.3 Agglomerative box plots for the Crockways site: a) categorised vegetation cover; b) water depth; c) original velocity; d) re-scaled velocity and e). sediment depth.

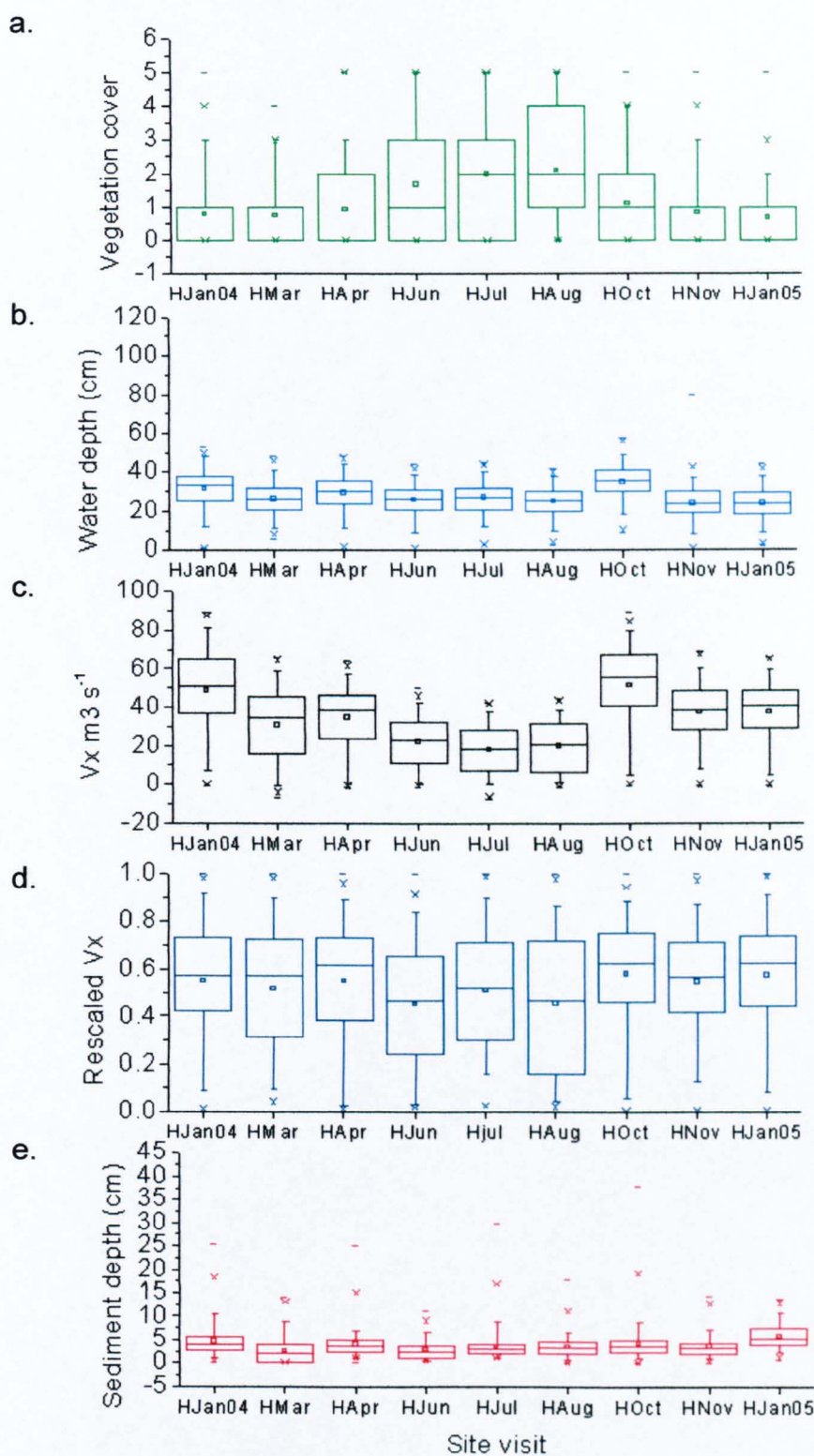


Figure 5.4 Agglomerative box plots for the Hydeclift site: a) categorised vegetation cover; b) water depth; c) original velocity; d) re-scaled velocity and e) sediment depth.

living vegetation: moribund vegetation remaining in the reach in August and October was regarded as still impacting on flow and was included as a contributor to total vegetation cover. The relative timing of vegetation growth and decline in the grid-reaches, and the abundance of vegetation in each season, are useful in informing other analyses in this and other chapters.

5.3.2 Agglomerate analysis: water depth

The water depth plots are shown in Figure 5.3b and 5.4b and indicate that water depth is greatly influenced by discharge, as indicated by the matching fluctuations in discharge, velocity and water depth at both sites. However, in general, seasonal cycles and storm events are less easily discerned in the water depth plots than in the velocity records, and average water depth at both sites is less sensitive to changes in discharge than is water velocity. There are also contrasting trends in the summer months at both sites, which are at odds with the velocity and discharge records. At Hydeclift, the water depth values are constant from June to August, in keeping with the constant velocities and discharge in this period, while at Crockways, constant velocities and discharge are also experienced but a staggered increase in water depth from June to August can be observed. As this bears no relation to discharge behaviour in this period, it is likely that vegetation influence has caused a change in the discharge/depth relationship. The influence of vegetation cover on water depth is not evident at Hydeclift where vegetation cover is much lower and water depth continues to fluctuate in line with small changes in velocity and discharge. If a vegetation influence is assumed, then average water depths in the Crockways grid may be compared for the three summer months to give a quantitative estimate of

vegetation induced depth increases. The increase in average water depth between June and August is 0.16 m: 0.04 m between June and July and 0.12 m between July and August. These estimates are inconsistent with the much smaller 0.01 to 0.03 m average increase in water depth observed in the stage/discharge relationship at Crockways in Chapter 4.

The Crockways grid-reach cannot be considered typical of the overall site; it was purposely chosen as a straight, relatively shallow, and abundantly vegetated sub-reach, with a diversity of plant growth form, located between meander bends and contrasts with the larger downstream reach utilised in the PT stage analysis which includes the meander bends. The PT reach has much more physical in-channel diversity and consequently has only low and patchy vegetation cover and a conspicuous lack of emergent macrophyte species. This contrast between the Crockways grid and PT-reach is reminiscent to the contrast between the Frampton and Crockways PT-reaches discussed in Chapter 4. Indeed, the Crockways grid-estimate of 0.16 m, is similar to the 0.12 – 0.17 m summer stage increase estimated for the abundantly vegetated PT-reach at Frampton. It may be that a critical biomass, needed for the emergence of vegetation effects, was achieved in the grid-reach at Crockways but not in the PT-reach. The extension of this theory is that the effects of vegetation on water depth are highly localised, and that connectivity of water depth within a vegetated river reach is not as strong as commonly thought. The Grid-reach lies only 2-3 m upstream of PT1, where PT stage measurements were taken, and this suggests that the PT cross-section is most strongly affected by the level of vegetation cover downstream of the cross-section, and not upstream. If vegetation is considered

as a semi-permeable dam (Champion and Tanner, 2000), then vegetation effects may reasonably be expected to propagate upstream more so than downstream.

It must be remembered, however, that the PT and grid estimates are not strictly comparable. The PT estimate from Chapter 4 compares average water depth changes over the entire summer period to the winter average over a large range of discharges, while the grid-survey estimate, by contrast, represents the average increase between two points in time; from low to moderate vegetation cover in June to maximum vegetation cover in August and over similar discharges. This difference in methods may effectively minimise the PT estimate and maximise the estimate based on the grid-data. However, methodological differences aside, the results do provide tantalising evidence of localised site differences at Crockways. This lack of connectivity in water depth is explored in more detail in later sections of this chapter, where water depth and velocity are considered over individual cross-sections in the grid-reach.

Water depth values for each site should show a similar distribution and structure between repeated surveys, regardless of discharge changes, providing no major morphological changes occurred between surveys, and providing discharge remains relatively constant during each survey. The constancy at each site is evident in the box-plots where the average water depth between surveys can be seen to fluctuate with discharge while the inter-quartile ranges remain similar. This is the case for all surveys at Hydeclift, and all but two at the Crockways site. The October survey, and to a lesser extent the January 2005 survey, at Crockways display a much wider range of water depths than any of the other 7 surveys at this site. This indicates either a

change in reach morphology or that discharge changed markedly over the time taken to conduct the grid-survey. That all other surveys, before and after October, present a similar distribution for water depth suggests that no major morphological changes took place during the one-year study period at either Crockways or Hydeclift. Discharge measurements from October (Figure 5.2) show a significant increase in the discharge between CS1 and CS20 and this indicates that the velocity values from this survey may contain patterns in water depth and velocity introduced by discharge changes.

5.3.3 Agglomerate analysis: stream-wise velocity

The longitudinal, or stream-wise, velocity (V_x) describes water movement in a direction parallel to the riverbank. The magnitude of the stream-wise velocity may be affected at any one time by roughness factors at three different scales: (i) catchment scale influences, e.g. velocity at each sample point will tend to increase with increasing discharge; (ii) reach scale influences such as local channel morphology, e.g. faster velocity across riffles and lower velocity in pools; and (iii) by smaller scale influences such as changes in channel substrate and bedform and the presence of in-channel macrophytic vegetation. Of the three scales of influence outlined above, the second scale, reach-scale morphology, could be deemed the least influential in this study. The grid-reaches were both chosen for their relatively uniform channel morphology (both sub-reaches possess a straight channel planform and lack major in-channel diversity), and this selection should minimise the influence of channel morphology on spatial variations in velocity and help isolate the influence of vegetation. Regardless of the specifics of local channel morphology,

morphological influence is likely to remain temporally constant over a one-year period and should not exert any significant influence when repeat surveys are compared.

Bedform influence is also largely absent from the grid-reaches, due to the dominant coarse substrate, which is non-cohesive and resists transport. Bedforms are, however, predicted to form from fine sediment deposits accumulated beneath vegetation stands in summer (Wolfert *et al.*, 2001). If local morphology, and vegetation-independent bedforms, can be discounted, then the two major influences on the point velocities at Crockways and Hydeclift will be: (i) temporal changes in discharge, including long-term seasonal trends and stochastic storm events; and (ii) the growth or decay of aquatic plants. The influence of vegetation may also be subdivided, whereby the plants provide physical obstacles to flow in terms of: (a) the plant mass; and (b) bedforms associated with vegetation such as 'obstacle bars' and 'chute channels' (Wolfert *et al.* 2001).

5.3.4 Rescaled V_x

It is evident from the agglomerative box-plots and from the discharge measurements (Figure 5.2) that the un-modified V_x values at both sites were highly discharge dependent. There is a suggestion of an underlying seasonal pattern, with low velocity values and low discharges in the summer months and high velocities in winter due to the higher discharges experienced during this period. This seasonal pattern may equally have resulted from vegetation-induced velocity reductions in summer, as considered in Chapter 4, but two pieces of evidence contradict this.

Firstly, the seasonal pattern in velocity is arguably more evident at Hydeclift than at Crockways, despite Crockways possessing the higher vegetation cover. And, secondly, the dominant influence of discharge is also expressed in the storm events that disrupt the seasonal cycle: in January, April and October 2004 and January 2005 at Crockways and October 2004 at Frampton. Any vegetation effects that may have occurred have been obscured in the raw data by the dominant influence of discharge.

Figure 5.3e and 5.4e attempt to minimise the influence of discharge by utilising re-scaled V_x values obtained through linear re-expression of the raw velocity data. This required that the smallest observed value in the raw data is set as the minimum value in the new scale, and that the largest observed value be placed as equal to the maximum value in the new scale. All other data velocity values are fitted between these boundary values and the relative distances between all values are maintained (Hartwig and Dearing, 1979). In this case, the V_x values were re-scaled to fit on a scale between 0 and 1 (known as z-scores), by subtracting the lowest value in the original data set from each other observed value, and by dividing the remainder by the range of the original values, i.e.:

$$V_{x\text{-re-scaled}} = \frac{(V_x - \text{min.})}{\text{range}} \quad (5.1)$$

Where: $V_{x\text{-re-scaled}}$ = re-scaled stream-wise velocity value

V_x = original stream-wise velocity value

min.= the minimum value in the original V_x data set

range = the range of values in the original V_x data set

(Hartwig and Dearing, 1979).

Re-scaling effectively removes the first order influence of discharge (i.e. the influence of discharge on mean velocity); the internal structure of each data set is unchanged, but the datasets are no longer differentiated in absolute terms, and this allows the inspection of the data sets as if they were taken at one constant discharge. Discharge will, however, still demonstrate effects in the variance and in the range of values. The removal of discharge events and cycles in mean velocity reveals the seasonal changes in V_x imposed by the growth and decay of vegetation. For example, the re-scaled data exhibit lower velocities in summer than in winter: from an average of 0.63 in April to 0.37 in August at Crockways, and an average of 0.55 in April to 0.45 in August at Hydeclift. These mean values indicate that the seasonal velocity pattern is stronger at Crockways, and less marked at Hydeclift, in accordance with differences in vegetation cover. Site differences are also shown in the general pattern of the box-plots; the Crockways site exhibits a smooth and gradual shift from a winter velocity regime to a summer velocity regime, while the summer velocity regime at Hydeclift is shorter and more abrupt. The seasonal re-scaled velocity pattern at Crockways is almost the exact inverse of changes in vegetation cover and this provides support for the velocity cycle as being a by-product of vegetation growth. The velocity pattern at Hydeclift plantation bears less resemblance to the seasonal vegetation trends at the site.

The overriding influence of discharge in the original V_x data also served to obscure changes in the diversity of velocity values between surveys. In the original V_x data the inter-quartile range was greatly related to discharge. At higher discharges, e.g. April 2004 at Crockways, the box-plots displayed larger boxes and seemingly higher internal diversity. In the re-scaled data, we may expect to observe an increase in

velocity diversity in summer, as the growth of vegetation would be expected to introduce more physical diversity into the channel environment (Sand-Jensen and Mebus, 1996). Interestingly, this is the case at Hydeclift, where vegetation is less abundant, but not at Crockways, where vegetation cover is greater. The degree of variability at Crockways appears relatively constant throughout the year, except in August when the lowest degree of velocity diversity was observed, coinciding with maximum vegetation cover. The Crockways site shows an overall decrease in mean summer velocity relative to winter, and this can only be achieved under two scenarios, either: (i) un-vegetated areas of faster flow are outnumbered by vegetated areas of reduced flow; or (ii) velocity reductions within plant beds are of a higher magnitude than the increases in un-vegetated areas. Either way, the overall reduction in mean summer velocity suggests that spatial diversity in velocity will be necessarily *lower* in summer. At Hydeclift, a seasonal reduction in average velocity is not evident and, therefore, more diversity in the channel may be expected.

5.3.5 Agglomerate analysis: sediment depth

The summary measures of fine sediment depth do not display any consistent seasonal trend between sites: the variation evident between individual surveys in winter is greater than the variation between summer and winter surveys. Given the expected summer retention of fine sediment beneath macrophytes described in the literature (Sand-Jensen and Mebus, 1996), and revealed by the PT results at Frampton (Chapter 4), it was expected that the depth of fine sediment would be significantly higher in summer than in winter. The sediment changes described in Chapter 4 were based on average results obtained over two water years and were able to smooth the

effects of varied discharge conditions to distil the average seasonal trends. The grid measurements, in contrast are ‘snap shot’ measurements which were highly vulnerable to the influence of antecedent discharge conditions. For example, the lowest sediment levels at both Crockways and Hydeclift were recorded in March and in November 2004, following large flood events in mid February and mid to late October (Figure 5.1). These large flood events may have flushed sediment from the reach. Similar discharge dependency was reported by Cotton *et al.* 2006 and it is likely that periodic re-suspension of trapped sediment is necessary to maintain plant health; continuous sediment accumulation beneath the plants would mean that the plant becomes rooted in less and less stable substrate and is vulnerable to wash-out (Haslam, 1978) and the burial of plant material reduces photosynthetic ability.

Another possible explanation might be that the accumulation of fine sediments beneath plants in summer observed in previous studies (Sand-Jensen, 1998) may represent a shift in the *pattern* of sediment distribution and not a change in the absolute availability or retention of fine sediment. Welton (1980) suggested that a similar seasonal shift in the pattern of sediment retention occurred in Tadnoll Brook, a tributary of the River Frome: fine sediment accommodation in winter was mainly provided by the low velocity margins of the stream, with a change to temporary storage beneath macrophytes in summer. Schulz *et al.* (2002) and Wolfert *et al.* (2001) also report that retention of sediment within macrophyte beds may be balanced by an increase in erosion in non-vegetated areas. Spatial patterns of sediment retention and erosion, rather than absolute, reach-scale, fine-sediment depth, will be examined in Section 5.4.4 and Section 5.5.4.

5.3.6 Cross-stream and vertical velocity: diversity of the flow field

The analyses of stream-wise velocity (V_x) in Section 5.3.4 suggested that V_x is less diverse in summer at high biomass sites than in winter. It is possible that diversity in summer velocity may alternatively be channelled into changes in the *direction* of water flow and not flow-velocity. Cross-stream (V_y) and vertical water velocity (V_z) were recorded simultaneously with stream-wise velocity (V_x), and together make up the three orthogonal components of water flow. These components may be reconciled to provide a single resultant flow velocity in either the horizontal or vertical plane, and this allows the calculation of the angle of deviation from longitudinal flow.

Cross-stream velocity (V_x) distributions for four selected surveys (March, July August and November 2004) are shown for Crockways (Figure 5.5) and Hydeclift (Figure 5.6). These four surveys provide a seasonal contrast (at similar discharges (see Figure 5.2) and are complete (20 cross-sections) at both Crockways and Hydeclift. The V_x rose diagrams provide a summary view of the channel flow field as ‘from above’: an angle of deviation to the left of the baseline describes flow direction towards the left bank and a deviation to the right describes flow towards the right bank. In winter, the angle of flow direction is concentrated in a narrow range close to the 0° reference-line, indicating the general uniformity of flow-direction expected in two reaches that have a straight planform and lack significant physical in-channel diversity. If, as an arbitrary illustration measure, the majority of flow in each survey is said to be represented by the simple arbitrary measure of all groups with $n > 4$, then the relative diversity of dominant flow angles may be quantified

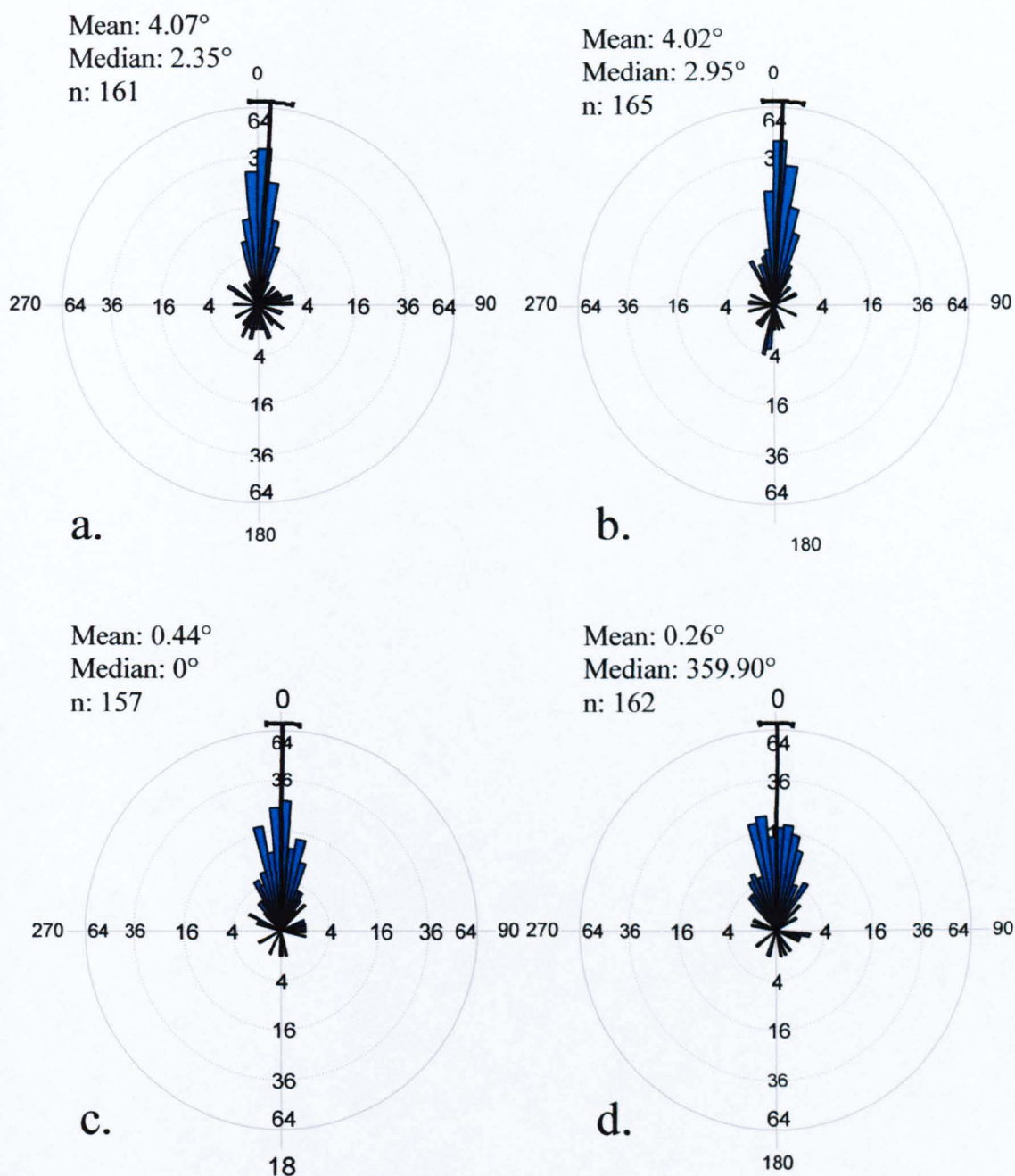


Figure 5.5 Rose diagrams (circular histograms) illustrating the relative diversity of flow orientation, in the horizontal plane, V_y , at Crockways in: a) March 2004; b) November 2004; c) July 2004; and d) August 2004. Observations are subdivided into groupings of 5° and the circular mean is indicated by a heavy black line.

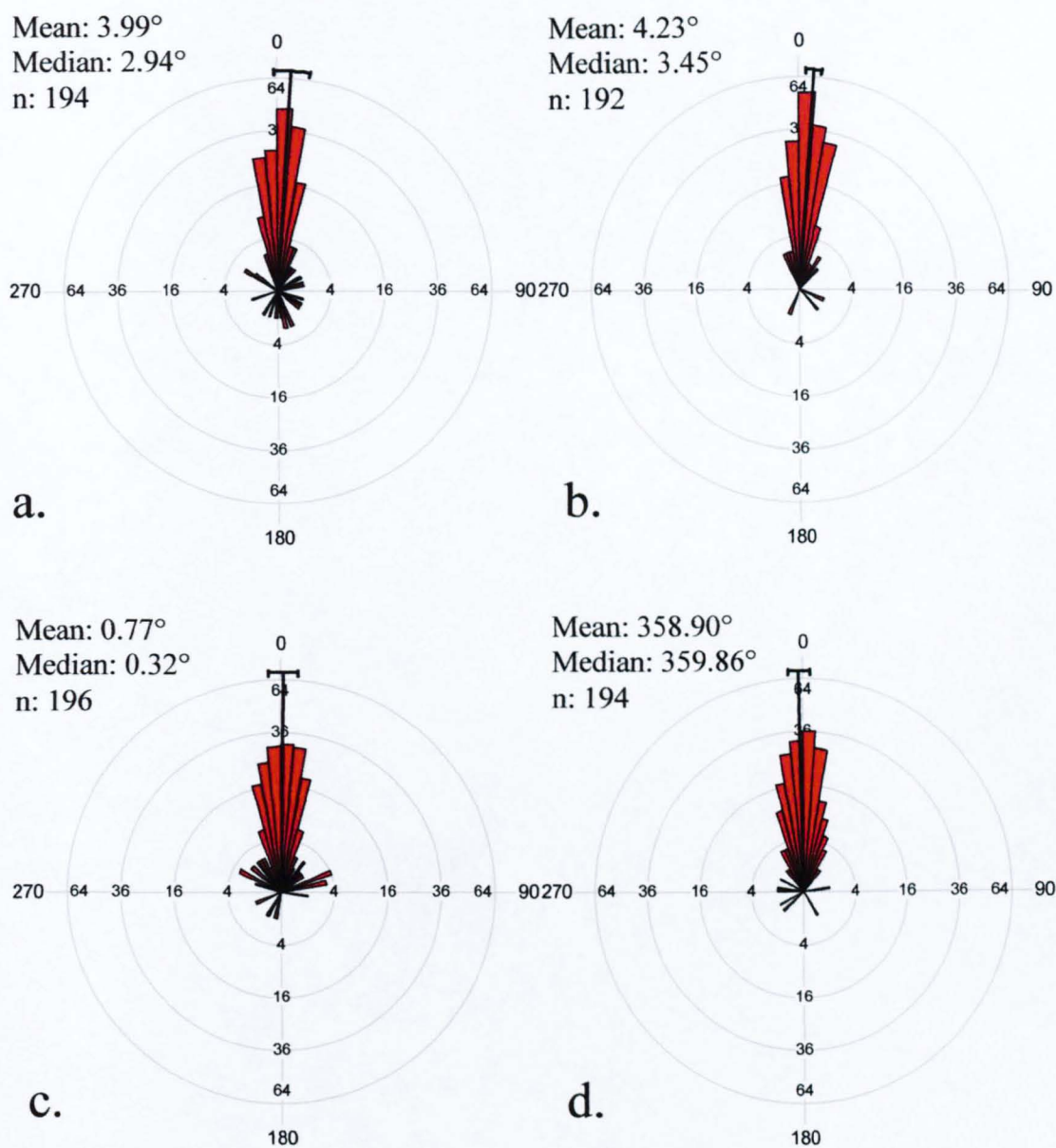


Figure 5.6 Rose diagrams (circular histograms) illustrating the relative diversity of flow orientation, in the horizontal plane, V_y , at Hydeclift in: a) March 2004; b) November 2004; c) July 2004; and d) August 2004. Observations are subdivided into groupings of 5° and the circular mean is indicated by a heavy black line.

between sites and between seasons. Seven groupings are found to have $n > 4$ at Crockways in March and six in November, while six groupings have $n > 4$ at Hydeclift in both March and November. As each grouping represents a 5° change in flow angle then this range can be quantified in degrees. In March at Crockways the majority of flow travels within -15° and $+20^\circ$ from the 0° reference line, while in November the majority of flow lies within -10° and $+20^\circ$. Both have a similar overall range of 35° and 30° . At Hydeclift, the majority of flow in March is within -15° and $+15^\circ$ of the 0° reference line, and in November within -10° to $+20^\circ$. The overall range for both the March and November surveys is 30° . These ranges indicate that, in winter, flow diversity in the horizontal plane is similar at both Crockways and Hydeclift.

Diversity in flow direction increased in the summer plots at both sites. The July and August plots at Crockways indicate that the majority of flow (all groupings $n > 4$) travels within the range of -30° and $+20^\circ$ in July and between $+30^\circ$ and $+35^\circ$ in August. The range at Hydeclift is somewhat smaller, -20° to $+20^\circ$ in July and 20° to 25° in August. This demonstrates that flow orientation, in the horizontal plane, is more diverse in summer than in winter and that this effect is more marked at Crockways than at Hydeclift. The average flow direction at both Crockways and Hydeclift also changes between summer and winter. In March and November average flow angle was $4.07^\circ - 2.96^\circ$ at Crockways and $3.99^\circ - 4.23^\circ$ at Hydeclift, while in summer no dominant flow direction may be observed at either site and the average flow angle is at or very close to 0° . This indicates that plant growth may act to override dominant morphological influences on cross-stream velocity.

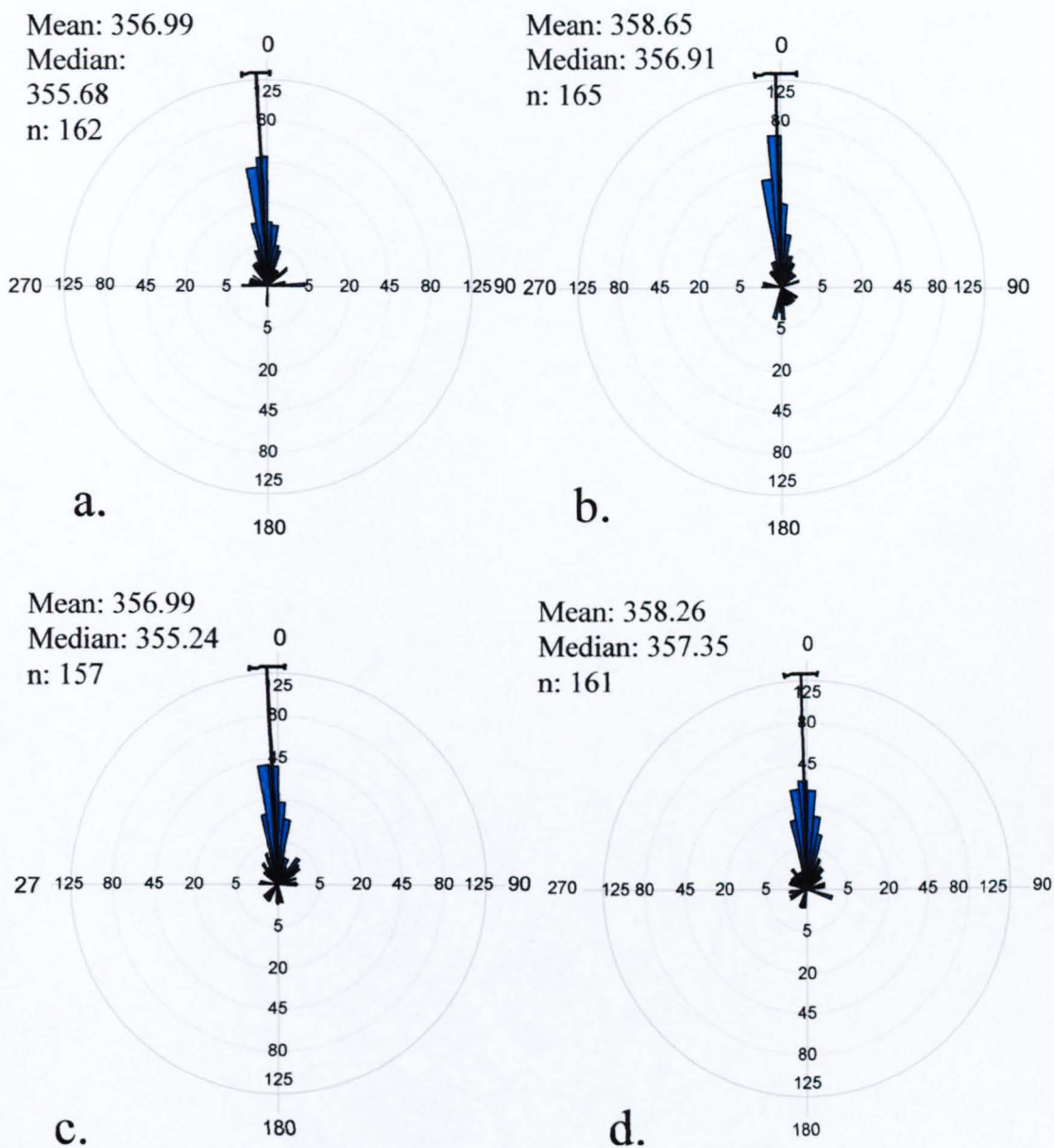


Figure 5.7 Rose diagrams illustrating the relative diversity of flow orientation, in the vertical plane, V_z , at Crockways in: a) March 2004; b) November 2004; c) July 2004; and d) August 2004. Observations are subdivided into groupings of 5° and the circular mean is indicated by a heavy black line.

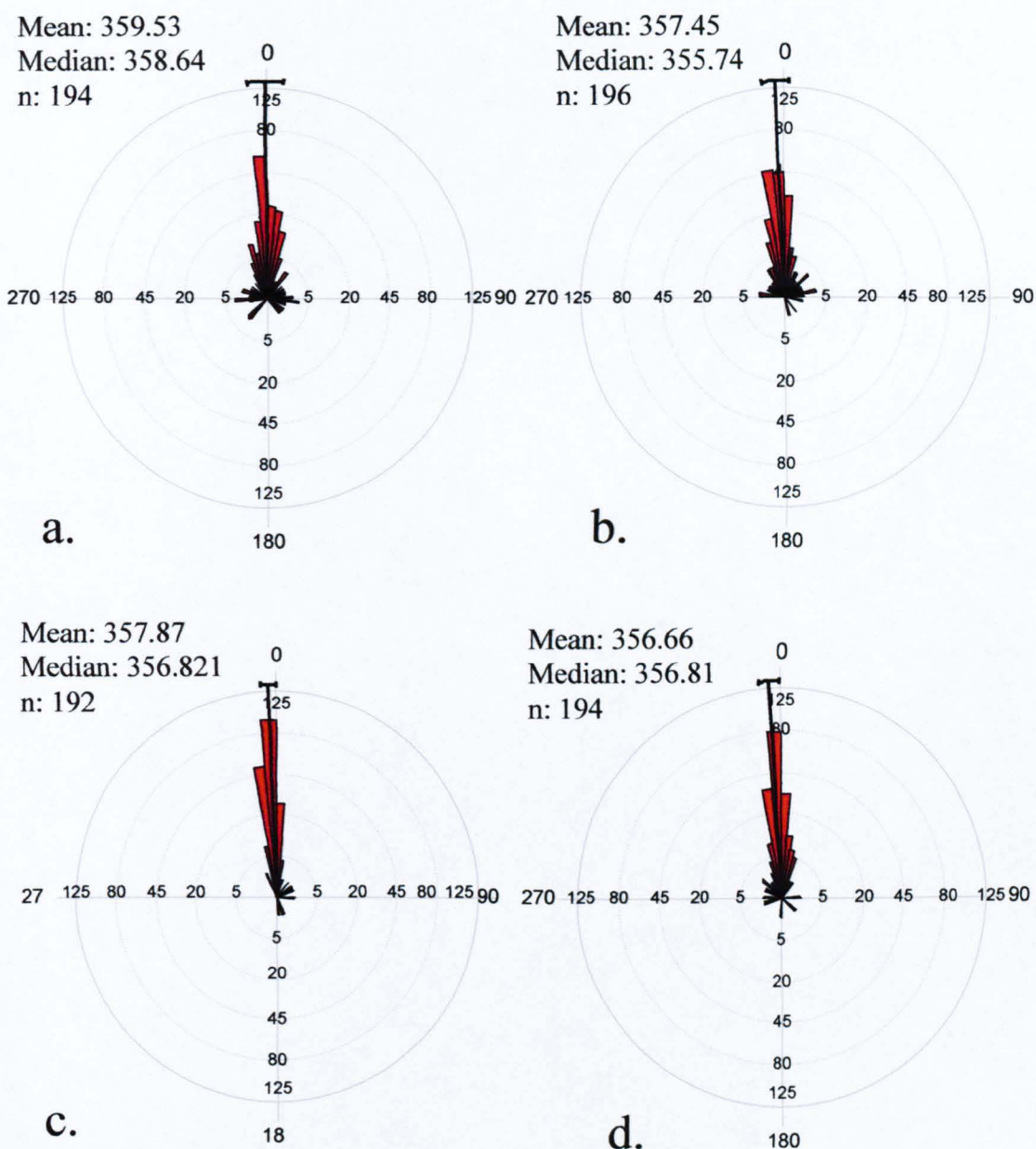


Figure 5.8 Rose diagrams illustrating the relative diversity of flow orientation, in the vertical plane, V_z , at Hydeclift plantation in: a) March 2004; b) November 2004; c) July 2004; and d) August 2004. Observations are subdivided into groupings of 5° and the circular mean is indicated by a heavy black line.

Figure 5.7 and Figure 5.8 illustrate the vertical velocity (V_z) distributions for four selected surveys (March, July August and November 2004) for Crockways and Hydeclift. The V_z rose diagrams act as a summary longitudinal slice through the channel: an angle of deviation to the right of the baseline indicates flow direction upwards towards the water surface and an angle of deviation to the left describes downward flow towards the channel bed. A comparison of the summer and winter rose diagrams does not reveal any systematic change between the winter and summer V_z distributions at either field site: the change apparent between the winter and summer distributions is no larger than observed between the distributions in March and November and between the distributions in July and August. This suggests that macrophyte growth has a greater effect on horizontal flow direction than vertical flow direction. However, this summary analysis does not exclude there being a change in the spatial *patterning* of V_z .

The results from the cross-stream (V_y) analyses illustrate that cross-stream flow diversity, or diversity in flow *direction*, is greater at both sites in summer than in winter and that this effect is greater at the higher biomass Crockways site. By contrast, the V_x analyses in Section 5.3.4 suggested that diversity in streamwise velocity, magnitude of velocity, was greatest in winter at the high biomass Crockways site and greatest in summer at the low biomass Hydeclift site. Perhaps the increased flow diversity predicted in vegetated channels in summer (Allan, 1995; Holmes, 1999; Large and Prach, 1999; Champion and Tanner, 2000) is channelled into flow *direction* and not velocity *magnitude* at high biomass sites.

5.4 Segregated analysis: multiple cross-sections

In the second scale of analysis, selected surveys at each site were sub-divided into cross-sectional slices to help confirm patterns in the agglomerative data and to analyse seasonal changes in water depth and sediment depth in more detail. Four surveys were chosen at each site (March; July; August; and November 2004); these include only complete surveys of 20 m length and were selected to provide a summer/winter contrast. The multiple cross-sectional data are also presented in a different way to the previous section: the box-plots within each diagram are still separated by variable, but no longer contain seasonal data. Instead, each box-plot contains all the segregated cross-sections for each variable in one individual survey. The plots are shown in Figure 5.9 to Figure 5.16 and span four pages. Diagrams have been grouped by site, and by season, for ease of comparison.

5.4.1 Multiple cross-sections: vegetation cover

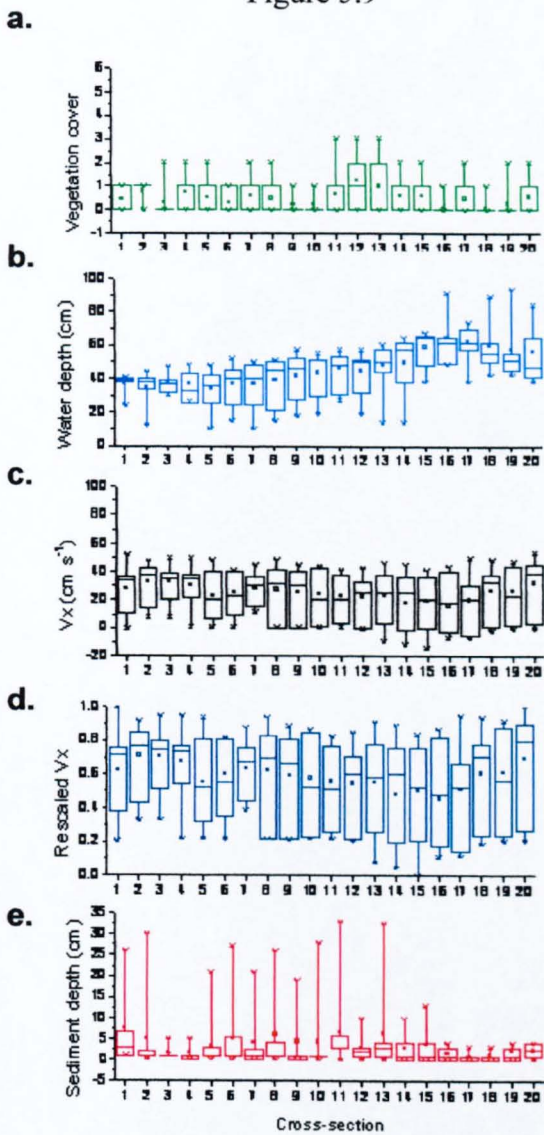
At both field sites, the lowest vegetation cover was recorded in March and this survey may be used as a baseline by which to judge seasonal changes in the growth and spatial distribution of in-channel vegetation. The March plots at both Crockways and Hydeclift reveal uniformly low vegetation cover across the reach.

In July and August, the vegetation cover at Crockways increased greatly from the March minimum, but this increase is not equal in each cross-section. The highest vegetation cover in July is found at moderate water depth and high water velocity, CS1 – CS13, while significantly less vegetation is found in the deeper cross-sections

of CS14 – CS20. The high water depth, low water velocity and paucity of fine sediment in the scour pool (CS14 – CS20) means that the rooting and growth of submerged plants is restricted, and as the pool is formed around the roots of a tree, shading of the channel may also affect vegetation cover. The high variability in CS14 – CS20 indicates that high vegetation cover is present in shallow and unshaded areas of these cross-sections. In August at Crockways, the vegetation has increased markedly from July levels across the whole reach. This is most evident in CS14 – CS20 where the vegetation cover increased more than for the other transects, though cover here is still significantly less than for CS1 – CS13 where cover is approaching 75%. Field notes taken during the survey suggest that this increase in cover may be due to the extension of submerged plants from upstream and to the left. Vegetation cover at Hydeclift increased dramatically in July and August from the March baseline level but was significantly less than at Crockways and the dominant factor controlling vegetation abundance at Hydeclift is likely to be shading of the channel by tree cover on the left bank. The variation in vegetation cover between cross-sections does not appear to be linked to water depth, which gradually increases throughout the reach, or to the winter velocity pattern, which would have conditioned the establishment of the plants. It does, however, coincide with summer velocity patterns, with high velocity generally occurring in the same cross-section as low vegetation cover, but this association is more likely due to the effects of plants on water velocity and not the reciprocal. It is likely that variations in the vegetation data at Crockways are linked to uneven shading by the riparian trees.

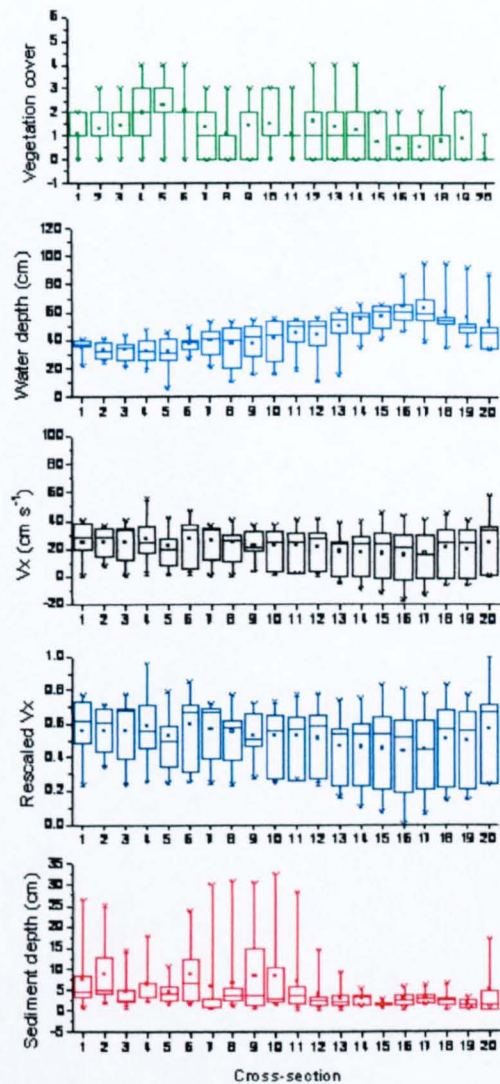
By November, the vegetation cover at Crockways and Hydeclift was dramatically reduced, relative to summer, but is still higher than the levels observed in March. of

Figure 5.9



March

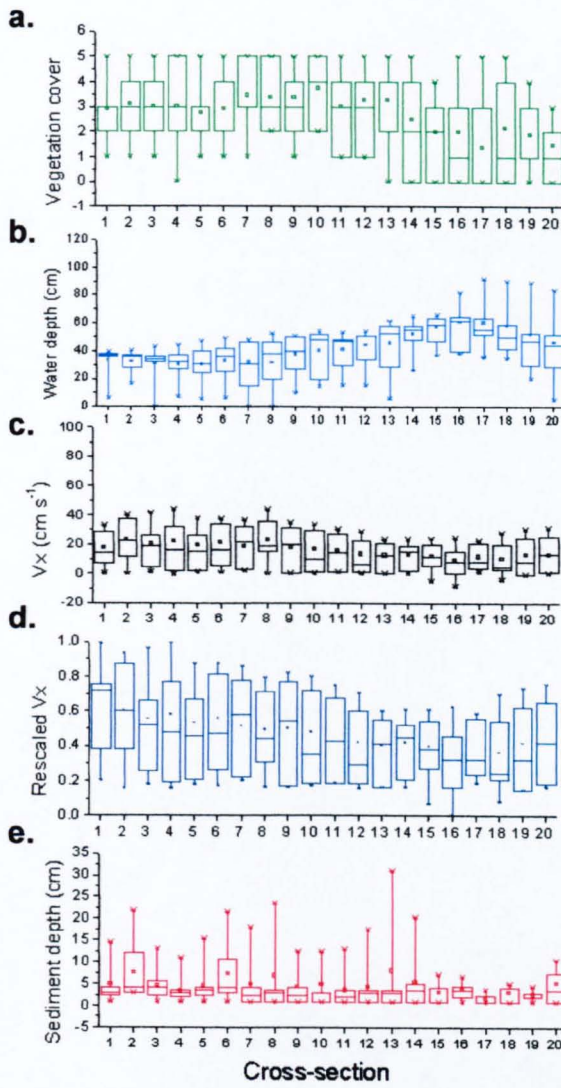
Figure 5.10



November

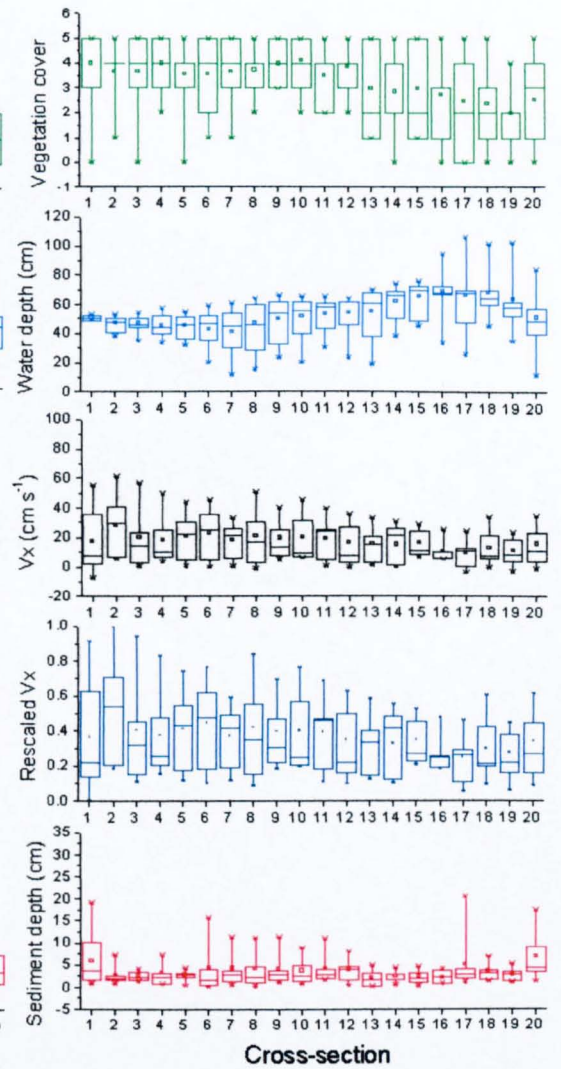
Figure 5.9 and Figure 5.10 Multiple cross-section box plots for the Crockways site in March and November 2004: a) categorised vegetation cover; b). water depth; c). original velocity; d) re-scaled velocity and e) sediment depth.

Figure 5.11



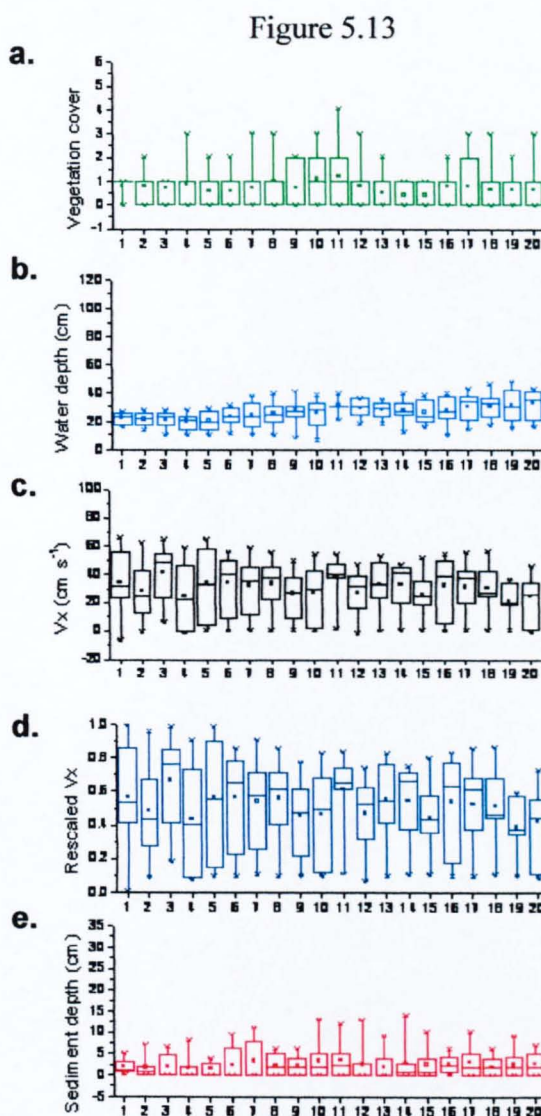
July

Figure 5.12



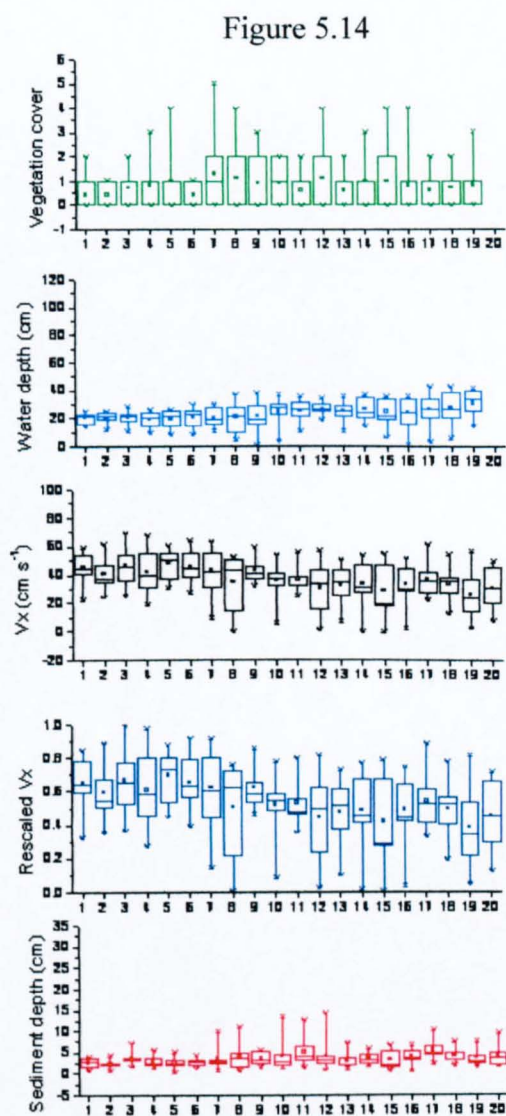
August

Figure 5.11 and Figure 5.12 Multiple cross-section box plots for the Crockways site in July and August 2004: a) categorised vegetation cover; b). water depth; c). original velocity; d) re-scaled velocity and e) sediment depth.



Cross-section

March



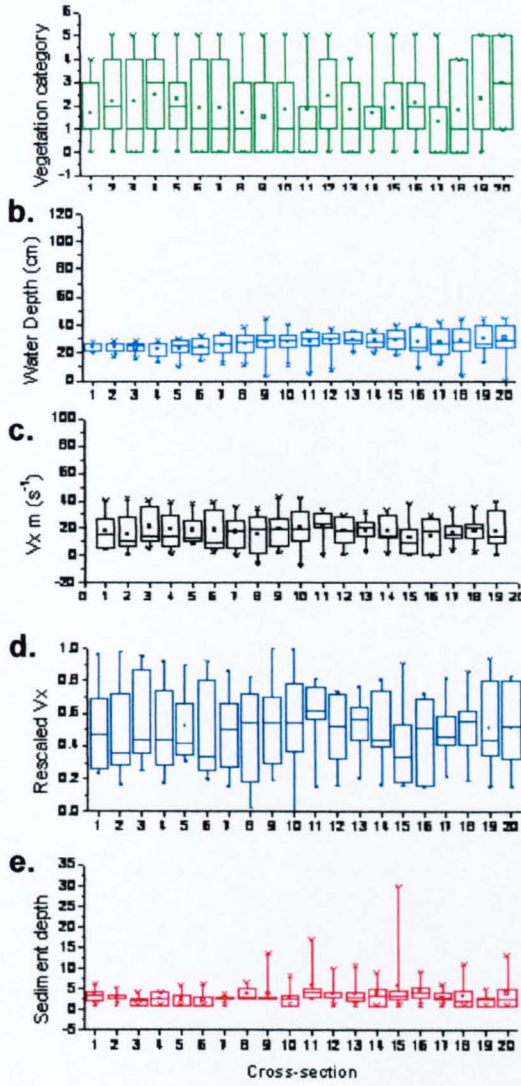
Cross-section

November

Figure 5.13 and Figure 5.14 Multiple cross-section box plots for the Hydeclift site in March and November 2004: a) categorised vegetation cover; b). water depth; c). original velocity; d) re-scaled velocity and e) sediment depth.

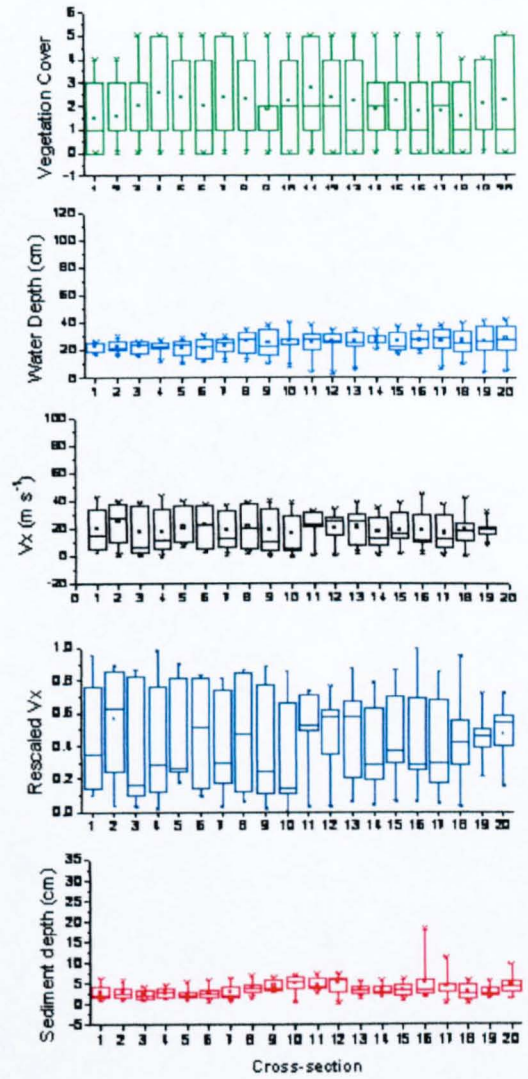
Figure 5.15

a.



July

Figure 5.16



August

Figure 5.15 and Figure 5.16 Multiple cross-section box plots for the Hydeclift site in July and August 2004: a) categorised vegetation cover; b). water depth; c). original velocity; d) re-scaled velocity and e) sediment depth.

the pool. These plants are rooted elsewhere in shallower water, but the long strands of the plant float near the surface of the deep pool. This is because dead vegetation still present in the reach was recorded in the vegetation totals. The pattern in the November vegetation cover is much less consistent than the summer pattern, and this may suggest the incremental senescence and washout of individual plants.

5.4.2 Multiple cross-sections: water depth

The channel at Crockways is deeper than at Hydeclift and becomes progressively deeper with increasing distance upstream. There is a fairly high degree of variability within each cross-section, and this reflects the cross-stream change in water depth from the channel margin to the channel centre. A comparison of mean, median and high-end outlier values adds more detail to this general picture and reveals the position of the pool in the Crockways reach: in CS16 – CS20, the high-end outliers follow the general downward trend of the mean values but are far removed from the mean and median values in terms of magnitude and indicates that these five cross-sections cross the pool at Crockways. There appears to be little difference in water depth patterns between the summer and winter surveys at Crockways, although, as in the agglomerate survey-plots, the water depth in August and July appear to have a higher absolute water depth, despite the lower velocities in these summer plots. This increase appears to be relatively constant across each cross-section, even in the pool cross-sections which have comparatively less vegetation cover, and this indicates that there is some connectivity in vegetation effects on water depth over short distances upstream.

At Hydeclift, the water depth plots reveal the overall uniformity of the channel. Little significant longitudinal variation is evident in the reach, though there is a slight and gradual increase in water depth from downstream to upstream, from CS1 – CS20. The variation across channel, from the margins to the channel centre also is less varied than at Crockways. Like at Crockways, there is little variation in water depth patterns from winter to summer; unlike at Crockways, however, the August data at Hydeclift are no higher in absolute terms than the other three surveys.

5.4.3 Multiple cross-sections: stream-wise velocity

The velocity plots at Crockways in winter directly mirror the water depth plots at the site. A scatter plot for each cross-section for March and November are shown in Figure 5.17, and illustrate the strong relationship between average water depth and velocity. The plots suggest a split in the data: CS1 – CS14 exhibit a different relationship for water depth and velocity to CS15 – CS20. CS15 to CS20 are the six deepest cross-sections in the reach, and though CS15 was not identified as a ‘pool cross-section’ (in Section 5.4.2), it may exhibit similar characteristics to the pool cross-sections.

In July, the same pattern is still largely evident but the difference between the lowest and highest velocity cross-sections has reduced, indicating that the vegetation may have modified or dampened the morphological signal present in the data (Figure 5.15). Given the disparity in vegetation cover in July, it is possible that the preferential growth of plants in the shallower, faster, cross-sections may have reduced velocity to values more comparable to those experienced in the less densely

vegetated, deeper and slower pool areas. Conversely, low vegetation cover in the pool may have allowed winter velocity characteristics to be maintained. In so doing, vegetation growth may have equalised velocity throughout the reach. Figure 5.17 illustrates this phenomenon and shows the altered depth/velocity relationship for the reach: the July plot shows an overlap between the two previously separated data groupings. In August reach-scale variability in average velocities has increased relative to July. This suggests that a further change in the depth/velocity relationship may have occurred and this is evident in the scatter plot in Figure 5.17, which shows less overlap between the data groupings and suggests that the reach is much less uniform than in July. This may indicate that late increases in vegetation abundance in CS15 – CS20 have equalised vegetation cover in the reach and have allowed the morphological signal to partially reassert itself.

The scatterplot for Hydeclift (Figure 5.18) suggests that the water velocities in winter are not significantly correlated with depth and this may reflect the physical uniformity of the channel. The scatterplots do not suggest a systematic change from winter to summer.

5.4.4 Multiple cross-sections: sediment depth

The sediment data illustrated in Figure 5.3 and Figure 5.4, indicated that no seasonal trend was captured by the agglomerative sediment data at either site. This is inevitably replicated by the absolute levels in the cross-section data, but the cross-sectional analysis reveals seasonality in the sediment *variability* across the grid reach between summer and winter surveys. In general, sediment may be observed to be

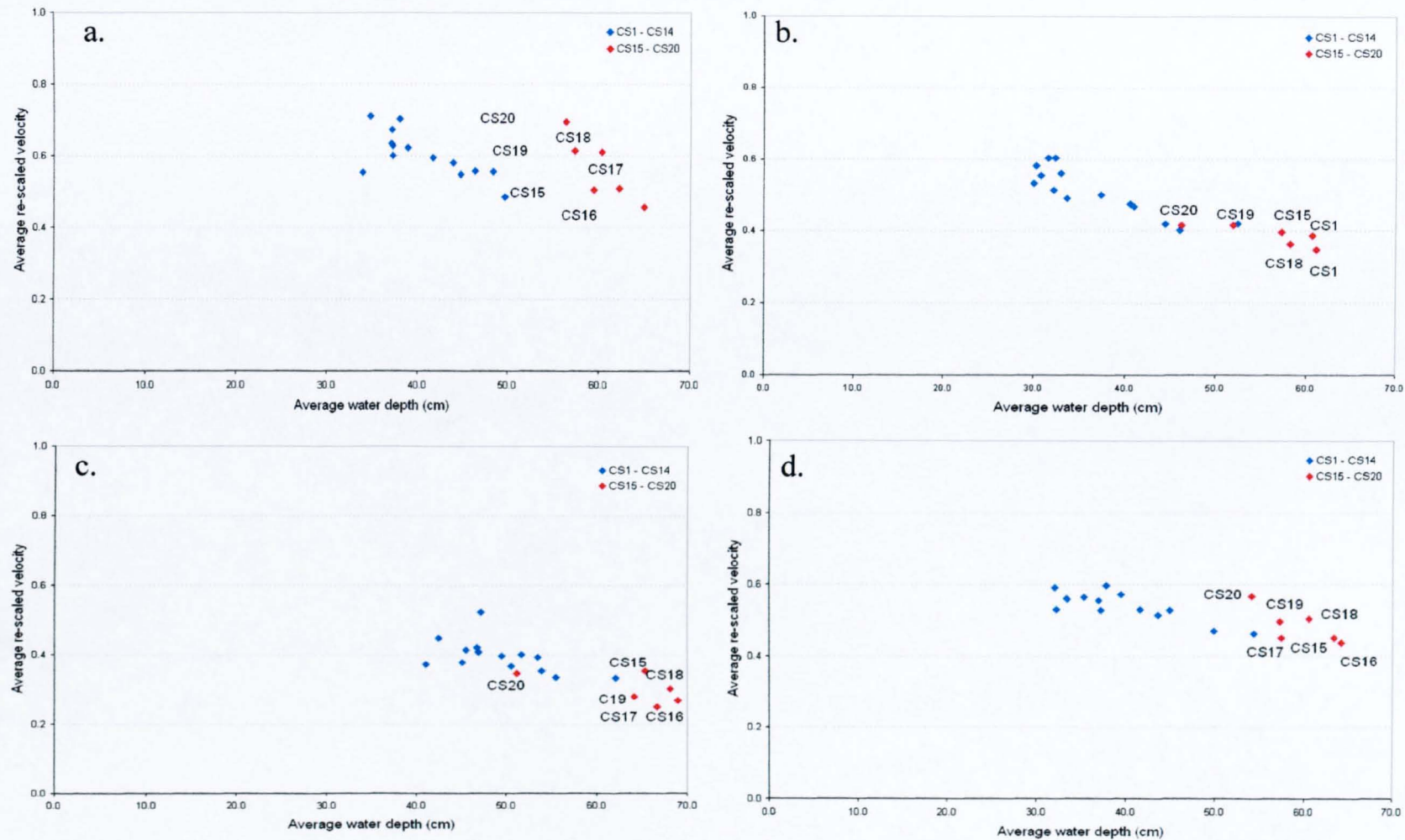


Figure 5.17 Scatter plot showing the changing relationship between water velocity and water depth for cross-sections at the Crockways grid reach from winter to summer: a). March; b). July; c). August and d). November 2004.

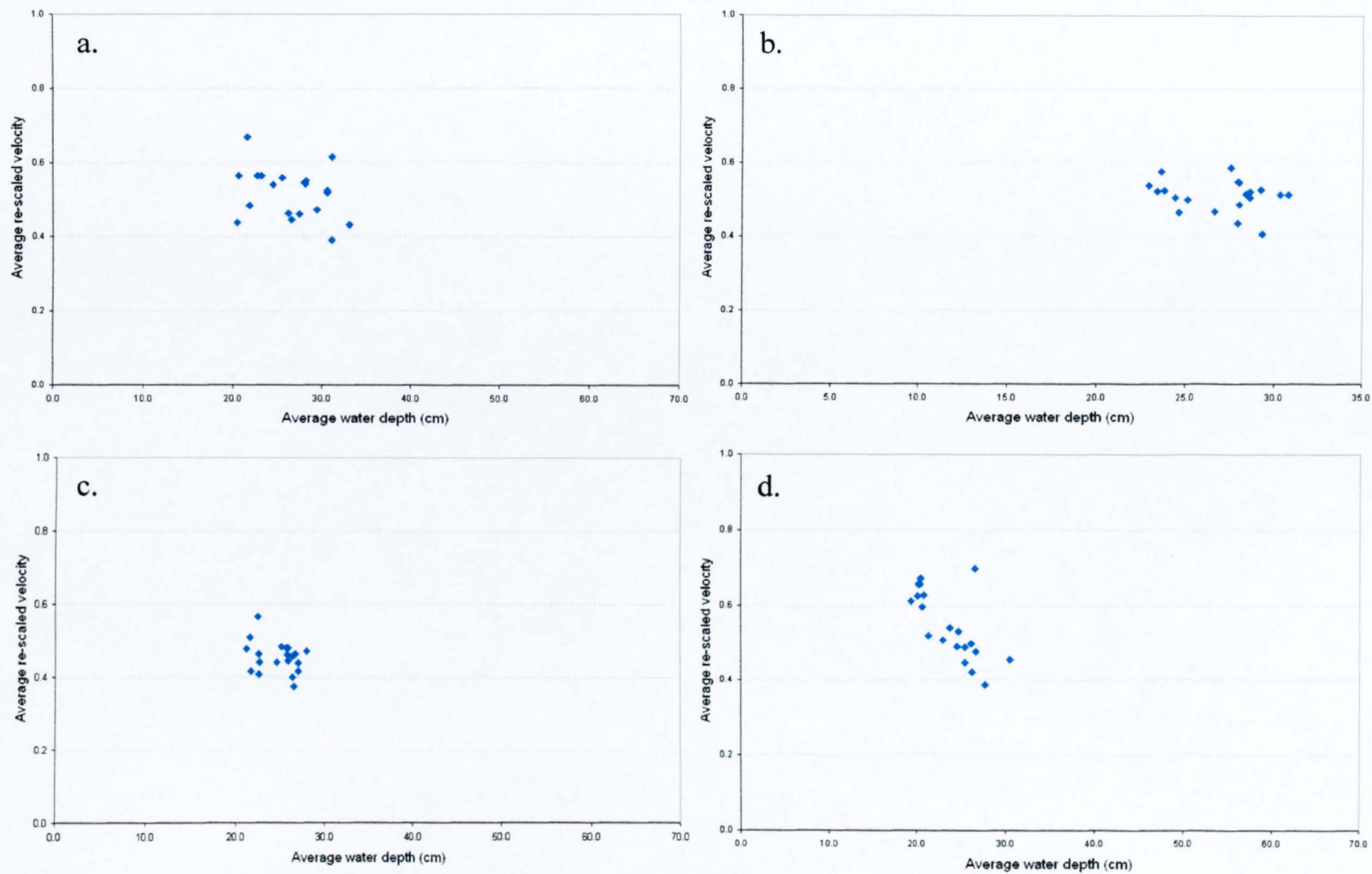


Figure 5.18 Scatter plot showing the changing relationship between water velocity and water depth for cross-sections at the Hydeclift grid reach from winter to summer: a). March; b). July; c). August and d). November 2004.

less evenly distributed across the channel in the winter months. This is evident in both the high variation in sediment depth between cross-sections and in the high variation within individual cross-sections, as indicated by the larger box sizes and the higher frequency and magnitude of outlier data in the March and November plots.

In July and August when the literature predicts maximum variation in sediment depth between vegetated and un-vegetated areas the cross-section values are actually the most uniform. This may suggest that sediment is predominantly stored in the channel margins in winter, with little storage in the central channel, while in summer this sediment may have been redistributed from the channel margins to the centre of the channel. However, this pattern of increased similarity between and within cross-sections in summer does not appear to be replicated at Hydeclift. The highest variation between cross-sections and the highest incidence of outliers occurs in July, and though August exhibits less variation between cross-sections the magnitude of outliers is higher in August than in the winter surveys. This trend suggests that the summer redistribution of sediments is not as evident at Hydeclift as at Crockways. This may be due to the lower availability of fine sediment at Hydeclift overall, but may also be because vegetation cover is much lower and less evenly distributed and the effects on sedimentation will be much less.

However, as noted in Section 5.3.6, the variation in sediment retention between surveys may be discharge controlled and the contrast between the winter and summer surveys used here may describe the differences between settled discharge conditions in summer and high flood flows in winter.

5.5 Segregated analysis: longitudinal slices through the grid-data

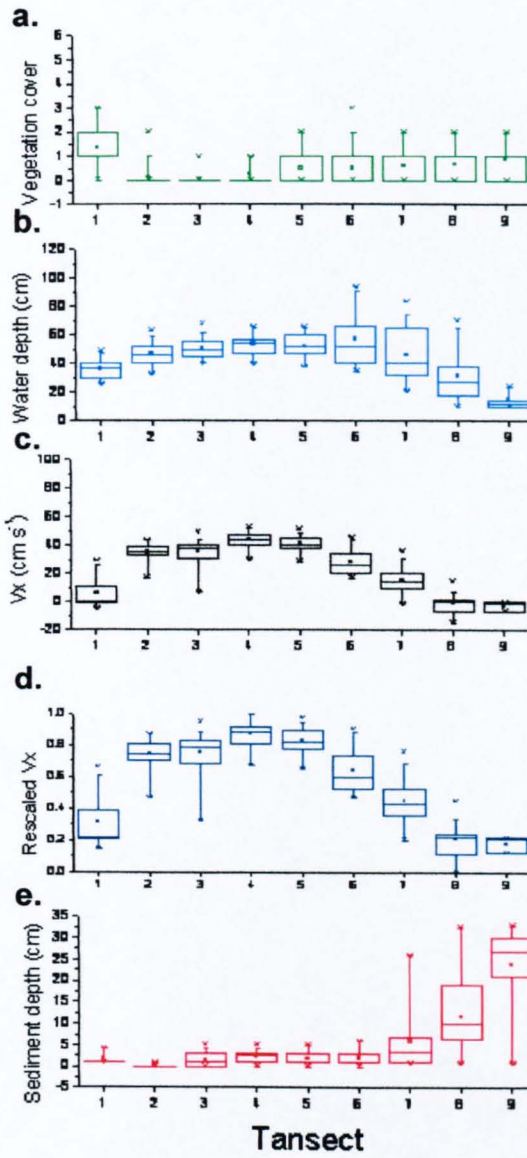
The final descriptive analysis divides the grids by transects, i.e. longitudinal slices parallel to the riverbank. This longitudinal sampling allows consideration of varying vegetation influence in different in-channel environments, e.g. channel margin vs. central channel. Again, the surveys from March, July, August and November 2004 were chosen for analysis. Plots are shown in Figures 5.19 to Figure 5.26 and have again been grouped by site, and by season, to aid comparison.

5.5.1 Transects: vegetation cover

Vegetation cover at both sites is lowest in March and this provides a baseline by which to judge the other three surveys: vegetation is generally uniformly low across both sites, though a slightly higher vegetation cover is recorded at the channel margins and this represents overhanging riparian vegetation.

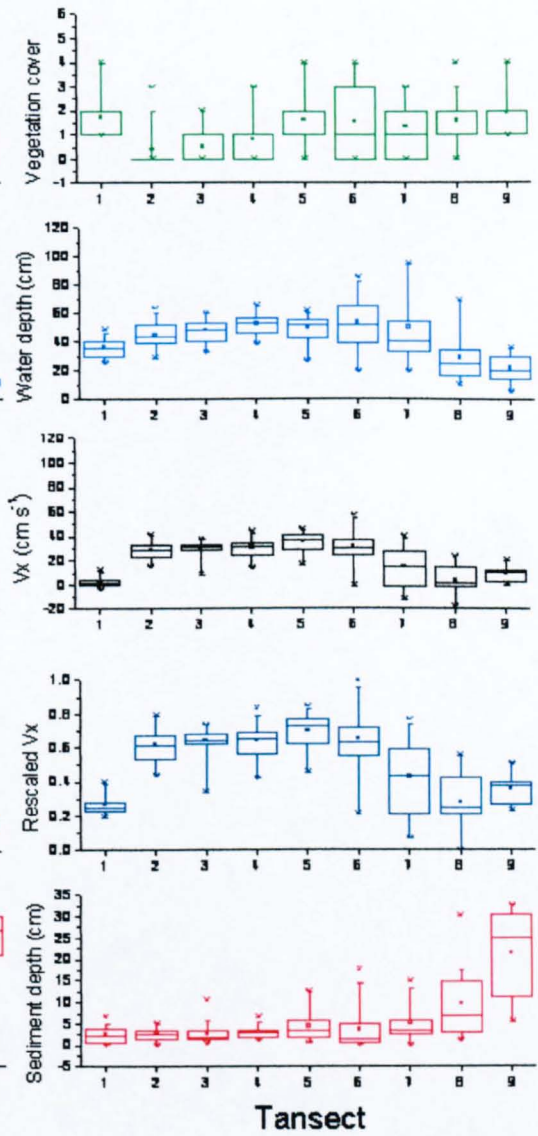
The July and August surveys have the highest vegetation cover and reveal a varied pattern of cross-channel vegetation growth. The vegetation patterns at Crockways are dominated by the physical controls of water depth or winter velocity. The variability in the data represents three overlapping physical habitat preference curves: (i) emergent species at the left bank; (ii) submergent species in the channel centre and (iii) emergent species at the right bank. Moving left to right at Crockways, T1 and T2 are close to the vertical left bank, and illustrate a decline from medium vegetation cover, dominated by overhanging riparian plants, to a channel minimum where high water depth and water velocity are unsuitable for the growth

Figure 5.19



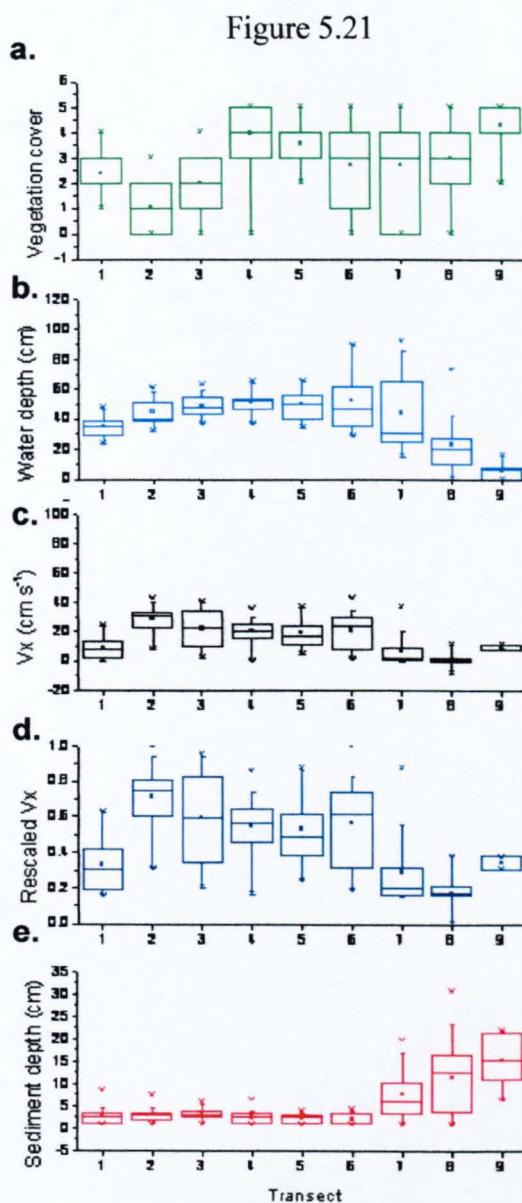
March

Figure 5.20

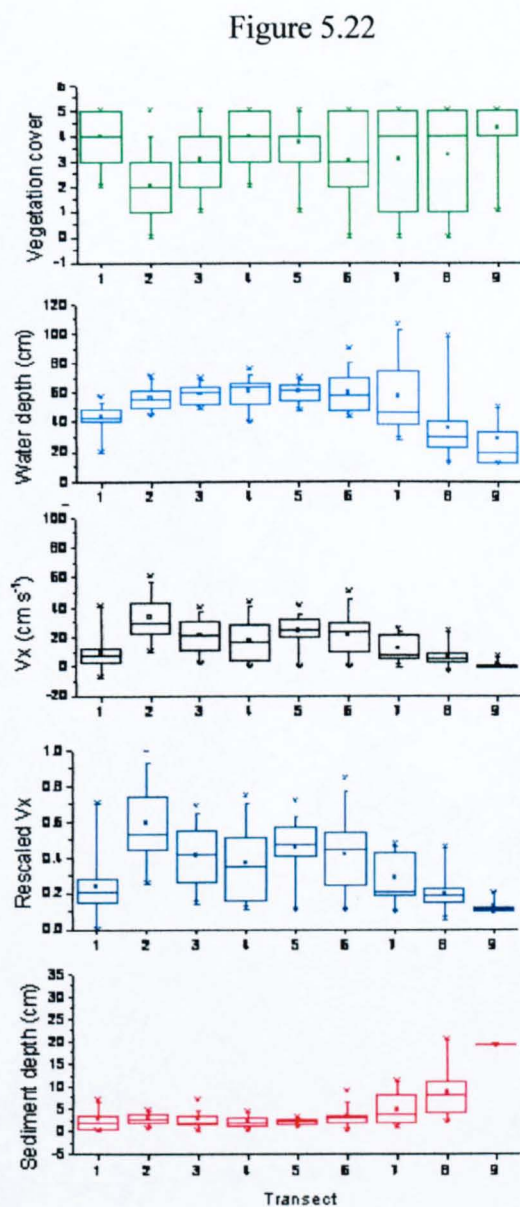


November

Figure 5.19 and **Figure 5.20** Transect box plots at the Crockways site in March and November 2004: a) categorised vegetation cover; b). water depth; c). original velocity; d) re-scaled velocity and e) sediment depth, plotted from the left bank to right bank.



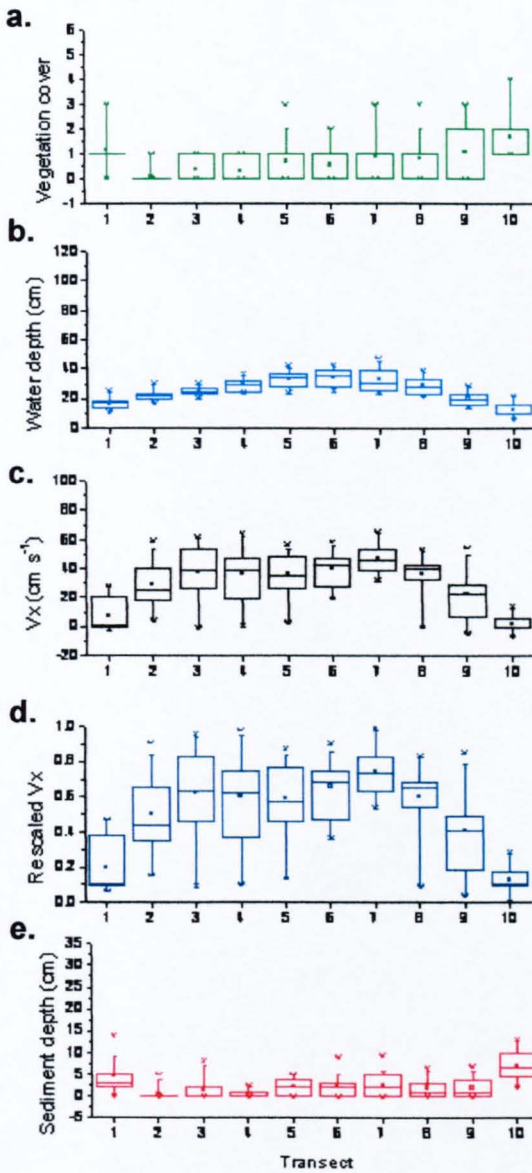
July



August

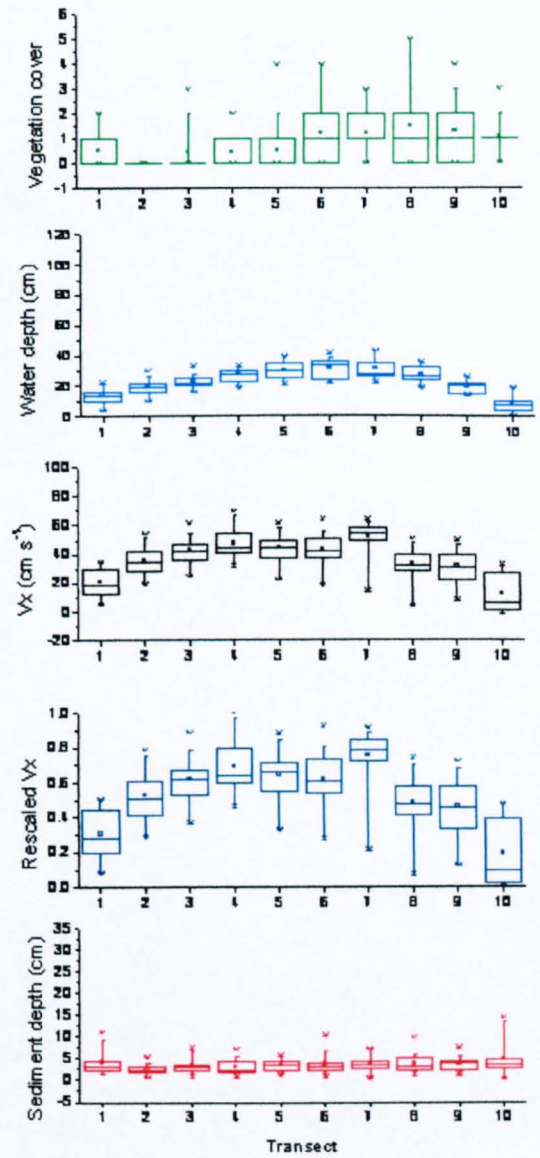
Figure 5.21 and Figure 5.22 Transect box plots at the Crockways site in July and August 2004: a) categorised vegetation cover; b). water depth; c). original velocity; d) re-scaled velocity and e) sediment depth, plotted from the left bank to right bank.

Figure 5.23



March

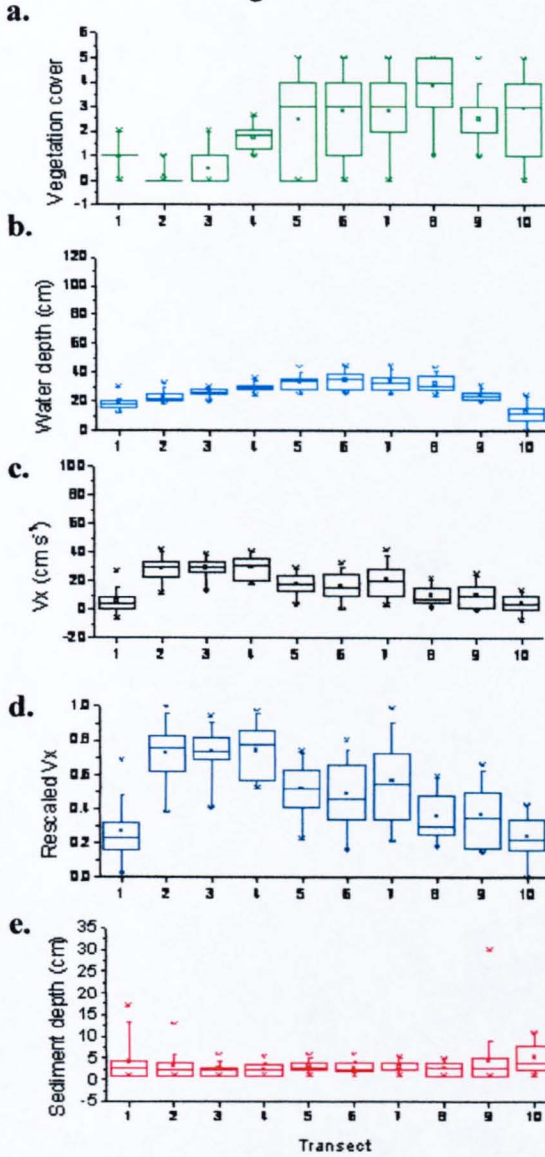
Figure 5.24



November

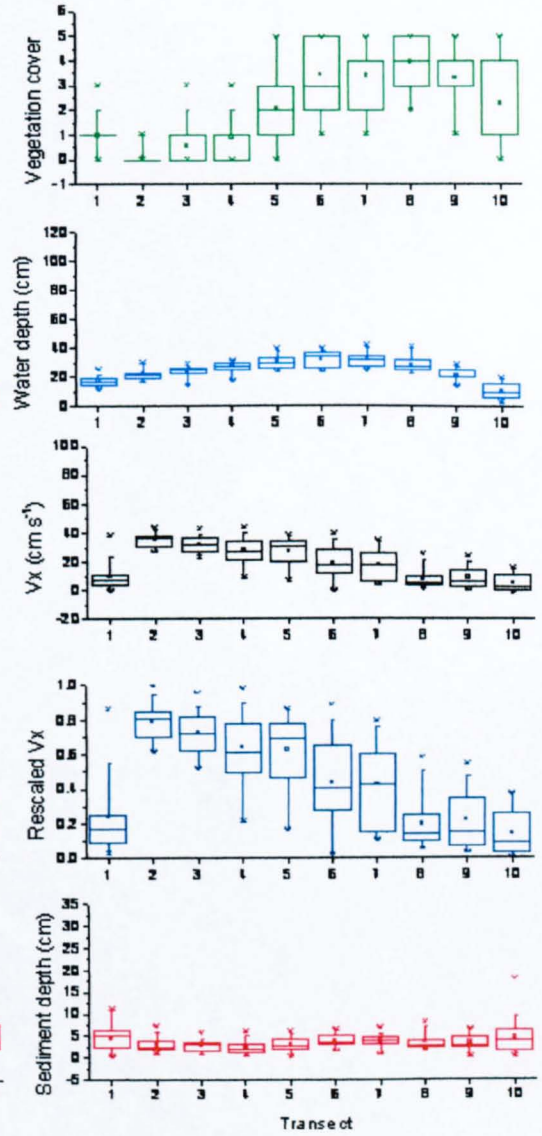
Figure 5.23 and **Figure 5.24** Transect box plots at the Hydeclift site in March and November 2004: a) categorised vegetation cover; b). water depth; c). original velocity; d) re-scaled velocity and e) sediment depth, plotted from the left bank to right bank.

Figure 5.25



July

Figure 5.26



August

Figure 5.25 and Figure 5.26 Transect box plots at the Hydeclift site in July and August 2004: a) categorised vegetation cover; b). water depth; c). original velocity; d) re-scaled velocity and e) sediment depth, plotted from the left bank to right bank.

emergent species and where shade is cast by the high vertical bank. The increase in vegetation cover in T3 – T4 represents an increase in submergent species as water depth increases and shade declines. Vegetation cover in T5 and T6 may still be considered high, but is somewhat less than in T4 and this is due to the relative decline in submergent plant cover as water depth and velocity declines towards the right bank. T7 is a transition zone between declining submergent and increasing emergent cover as the water becomes too sluggish and too shallow to support submergent species and instead favours emergent species. T8 represents only emergent vegetation, exclusively dominated by thick populations of *Sparganium erectum*, while T9 is also dominated by *Sparganium* but with a contribution from encroaching riparian species.

Summer vegetation patterns at Hydeclift differ markedly from the patterns displayed at Frampton in the same period. Shading of the channel by riparian tree cover on the left bank has severely restricted light availability and created a division between shaded and un-shaded areas. There is a distinct divide between T1 – T4 to the left of the channel and T5 – T10 to the right of the channel. The riparian trees cast their shade, and influence vegetation cover over more than half the channel width (~ 6 metres). Vegetation cover is equal across the un-shaded transects, T6 – T8, and this reflects uniform water depth across the channel. There is a reduction in vegetation cover in T9, where water depth and winter velocity increase slightly. T10 represents a thin bank of fine sediment and is the only transect environment capable of supporting emergent species at Hydeclift.

In November the vegetation cover at both field sites returns to much lower levels. The vegetation cover has not yet returned to the March minimum, and this is mainly due to dead and dying vegetation still remaining in the reach. Some live *Ranunculus* was still present, especially at Hydeclift, but both live and dead vegetation were later washed out by winter storms events.

5.5.2 Transects: water depth

Water depth at both sites, in both summer and winter, exhibits a general curved trend from bank to bank. The lowest depths are encountered at the channel margins, followed by a gradual increase in water depth towards the channel centre. At Crockways, the water depth plots record the vertical left bank, evidenced by the immediate high depth measurement in T1, contrasting with a more gently sloping channel profile, over a side-bar of fine sediment, at the right bank. T6 – T8 in the water depth plot describe the presence of the small pool at the upstream end of the reach; the pool is described by the larger inter-quartile range but has only minimum influence on average depth (Figure 5.20 to Figure 5.22). At the wider Hydeclift site, water depth is much lower than at Crockways and the contrast in depth from channel margin to channel centre is less evident in this more uniform reach. A vertical left bank is described by the immediate drop in water depth at T1, and this contrasts with a more gently sloping right bank. A small bank of fine sediment is present at the right bank in March, July and August but is not evident in the November plot (Figure 5.23 to 5.26). This bank forms under emergent *Petasides hybridus* (butterbur) and *Mentha aquatica* (water mint) vegetation which has declined by November. The

very small inter-quartile range in each transect reflects the uniformity of the reach and the absence of significant morphological diversity.

5.5.3 Transects: stream-wise velocity

The V_x transects display markedly different trends in winter and in summer. In winter the raw V_x and re-scaled V_x data follow the same general curved trend evident in the water depth plots: velocity is lowest at the shallow channel margins and highest in deeper water at the centre of the channel. At Crockways, marginal velocities are 71% lower than the higher in-channel velocities, and at Hydeclift this division is slightly more marked with a difference of 74%. The effect of water depth on winter water velocity is supported not just by the general trend, but also by site-specific distinctions. At Crockways, the vertical left bank and a more gently sloping right bank described by the water depth values is mirrored by velocity, which increases more rapidly at the left bank than from the right bank. Trends in water depth across the channel were shown in Section 5.5.2 to be very similar from winter to summer and this known trend can be used as a benchmark by which to investigate changes in velocity between seasons.

In summer, the curved bank-to-bank trend is severely disrupted at both sites. At Crockways the highest summer average velocity occurs 1.5 m from the left bank at T2, while the average re-scaled velocities in the two transects that experienced the highest winter velocities (T4 and T5) have decreased by 58% and 44% from March to August. The average velocity in T2 was also reduced between March and August, though by a smaller margin of 20 %. These figures reflect the overall decrease in V_x

at Crockways in August when the V_x pattern is more influenced by vegetation cover than by water depth. For example, T2 has both the highest V_x *and* the lowest vegetation cover. There is also an increase in V_x at T9 in July and this may describe a thin flow channel observed between the emergent vegetation and the bank line. This has disappeared by August following the encroachment of riparian vegetation. Scatter plots and for the transect-averaged velocity and water depth at Crockways are shown in Figure 5.27 emphasise the seasonal velocity changes. In March, a binary split in the data may be observed between ‘marginal’ and ‘central’ areas of the channel. In July, however, the data may be split into 3 groupings: marginal; central and ‘transitional’; and suggests a change in the velocity/depth relationship as vegetation growth alters the morphological signal. In August, the transitional phase is complete, perhaps signalled by a critical biomass, and the majority of the transect data plots as one data body. T2 is an obvious outlier in the August plot. This corresponds to the lowest vegetation cover in the reach and may represent a preferential flow channel (Gurnell *et al.* 2006). In November, the marginal/central split in the data has returned, though T6 may now be regarded as central, perhaps due to a slight decline in the width of the depositional side-bar at the left bank.

At Hydeclift, the summer relationship between water depth and V_x is severely disrupted. The highest average V_x has switched from T7 to T2 – T4, and this corresponds to the area of lowest vegetation cover. The re-scaled average velocity in the low vegetation transects *increased* by 26% between March and August, in marked contrast to Crockways, where velocity was reduced in even the fastest flowing transect. A scatter plot of re-scaled V_x and water depth is shown in Figure 5.28 and illustrates that differences exist between summer and winter at Hydeclift,

and also between Hydeclift and Crockways in each season. The scatter plot data for winter at Hydeclift are not split between marginal and central transects, and this reflects the uniformity of the reach. In July, there is an obvious split in the data between the shaded transects (T2 – T4) and un-shaded transects (T5 – T10). In August, this shaded/un-shaded split is accentuated: the shaded transects now include T5, following a decline in vegetation cover between July and August. The scatter plot also suggest a negative relationship between depth and velocity in the shaded transects, i.e. water velocity declines with increasing depth. This suggested negative trend may be explained by the location of the four transects (T2 – T5) along two gradients: (i) water depth, which increases from T2 – T5 in the centre of the channel; and (ii) vegetation cover, which also increases from T2 – T5 due to a progressive reduction in shading. It is apparent that the change in vegetation cover between transects imparts a stronger signal than the morphological influence of increased depth and distance from the bank.

5.5.4 Transects: sediment depth

The earlier agglomerative and cross-sectional analyses of sediment depth data showed that average sediment depth could not be linked to seasonal vegetation trends but that patterns of sediment variability throughout the reach might be linked to vegetation cover. The patterns in sedimentation found at the transect-level may help to investigate this second hypothesis.

The transect sediment data at Crockways demonstrated that there is a very obvious skew in sediment storage location in *both* winter and summer, and most sediment is

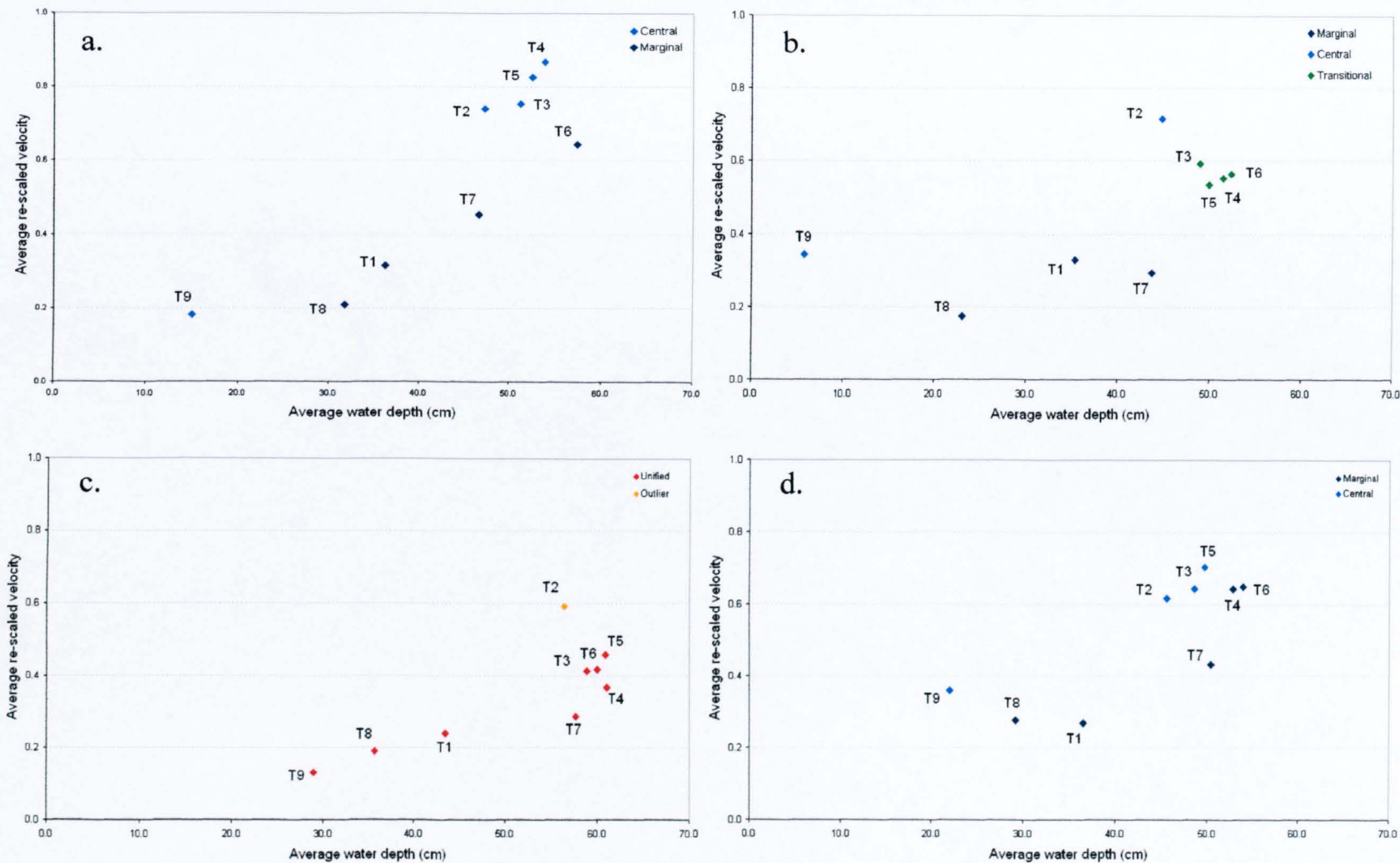


Figure 5.27 Scatter plot showing the changing relationship between water velocity and water depth for transects at the Crockways grid reach from winter to summer: a). March; b). July; c). August and d). November 2004.

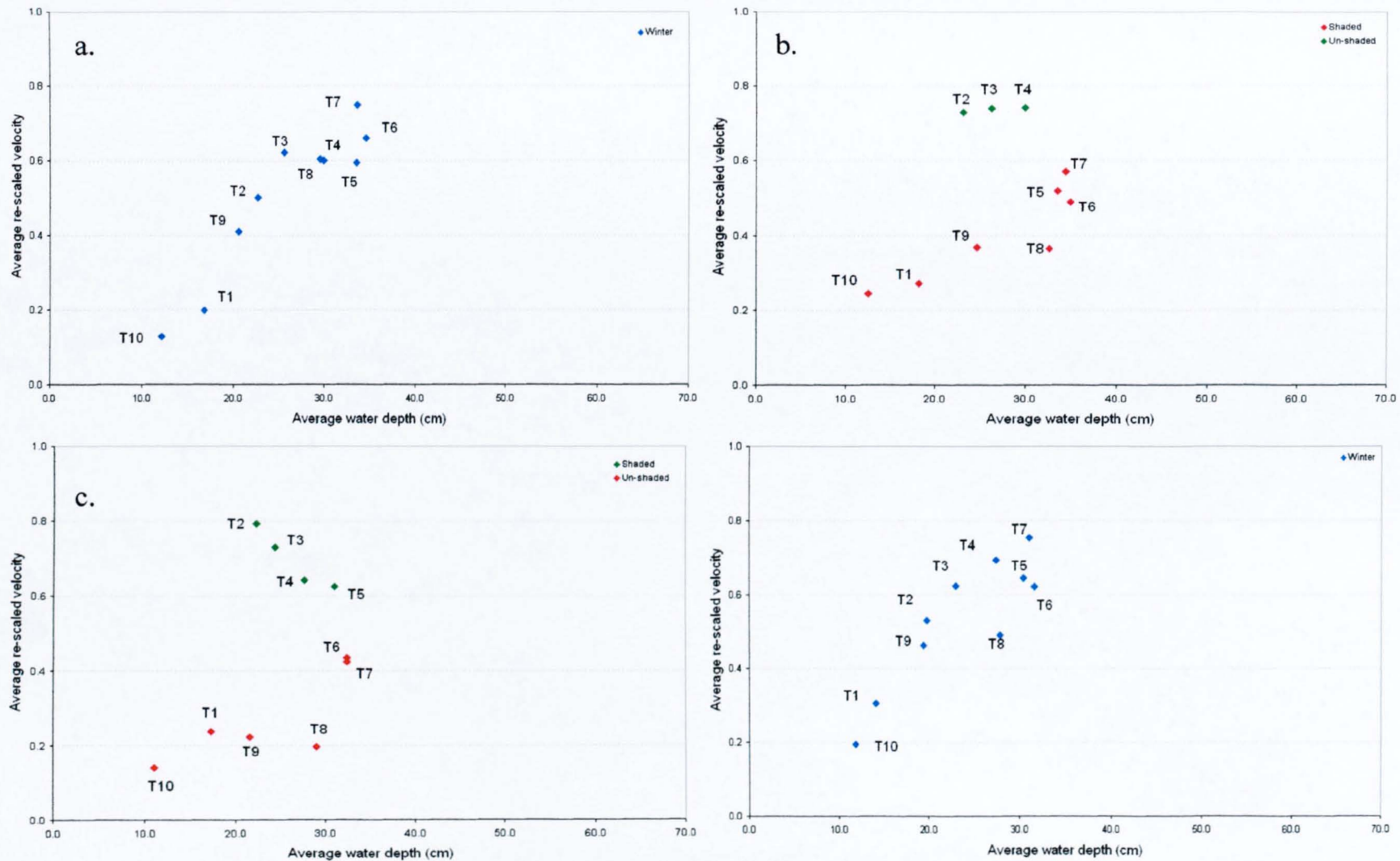


Figure 5.28 Scatter plot showing the changing relationship between water velocity and water depth for transects at the Hydeclift grid reach from winter to summer: a). March; b). July; c). August and d). November 2004.

stored in the final three transects to the right of the channel. This high average sediment depth in T7 – T9 describes the bar of fine sediment along the right-bank, while the extreme low outlier values describe the paucity of sediment in the pool environment at the top of the reach. The sediment bar can be seen to persist from summer to winter and the large perennial sediment deposits found here may act to obscure the smaller influence of the submerged plants. The bar itself, however, is also covered by vegetation and one would expect that sediment depth here would increase in summer relative to winter, but this is not the case. The highest average sediment depth in winter, in T9 is 24 cm in March and 22 cm in November, but only 15 cm in July and 22 cm in August. This reduction in marginal sediment levels in summer occurs despite the similar absolute sediment levels in the reach for all four surveys (Figure 5.3) and provides possible support for the redistribution of sediments from the margins to the channel centre in summer. However, an alternative explanation may be more accurate.

Sediment depth in each survey was only recorded in the wetted area of the channel and it may be that areas of deeper sediment recorded in winter were not submerged in the summer surveys and the number of measurements recorded at T9 in each survey may differ greatly. There are seven recordings in T9 in March and 11 in November, compared to five in July and only one in August. It may be that significant sediment storage was ‘lost’ between surveys, not because it had been physically removed or redistributed but because it had been effectively ‘written off’ by shifts in the wetted area of the channel. Given this situation, a perceived decrease in sediment storage at the margins may have obscured the increase in sediment in other areas of the bed in the summary and cross-sectional box plots. The field

surveys cannot be re-executed to include areas of the channel missed in summer but the three transects that represent the sediment bar may be removed from the analysis to reveal changes in sedimentation in the rest of the channel. Figure 5.29 shows seasonal agglomerate box-plots for each survey at Crockways, but where the data for T7 - T9 have been removed. The new plot is very similar to Figure 5.3 and reveals no overall seasonal trend. However, the reduction in sediment depth observed from June to August in Figure 5.3 is no longer evident and average sediment depth increases in these three months, perhaps reflecting increasing storage under vegetation. Figure 5.30 shows the transect data re-plotted without the data from T7 – T9. This does not appear to show any significant seasonal effects, i.e. the variation between summer and winter surveys is no greater than the variation evident between surveys in the summer season. It is recommended that full channel surveys and not just surveys of the wetted area of the channel be carried out in future surveys.

At Hydeclift, only a small marginal sediment bar exists, and this is only recorded in T10 of the survey. Nevertheless, the high sediment depth in this transect may obscure smaller seasonal changes in the other transects. In Figure 5.31 and 5.32 the data for T10 have been removed and the sediment data have been re-plotted at transect level for all surveys at agglomerate survey level to reveal any seasonal differences (Figure 5.31) and for each of the four sample surveys (Figure 5.32). Figure 5.31 differs very little from Figure 5.4 and no overall seasonal trend can be identified. Unlike at Crockways, there is no evidence of an increase in sediment depth beneath vegetation in the summer months. However, Figure 5.32 reveals that sediment is more equally distributed across the channel in winter than in summer. In July, sediment depth in T2 – T4 is noticeably lower than in T1 and T5 - T8. T1 has

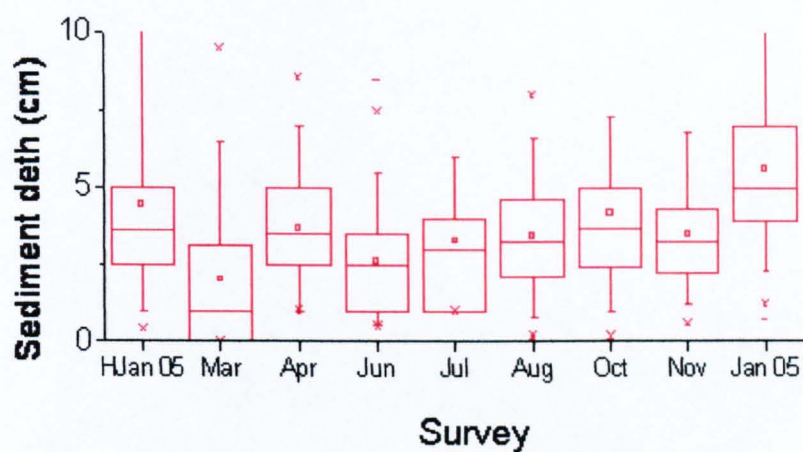


Figure 5.29 Agglomerative box plots of sediment depth at Crockways re-executed for Transects 1 – 6 only.

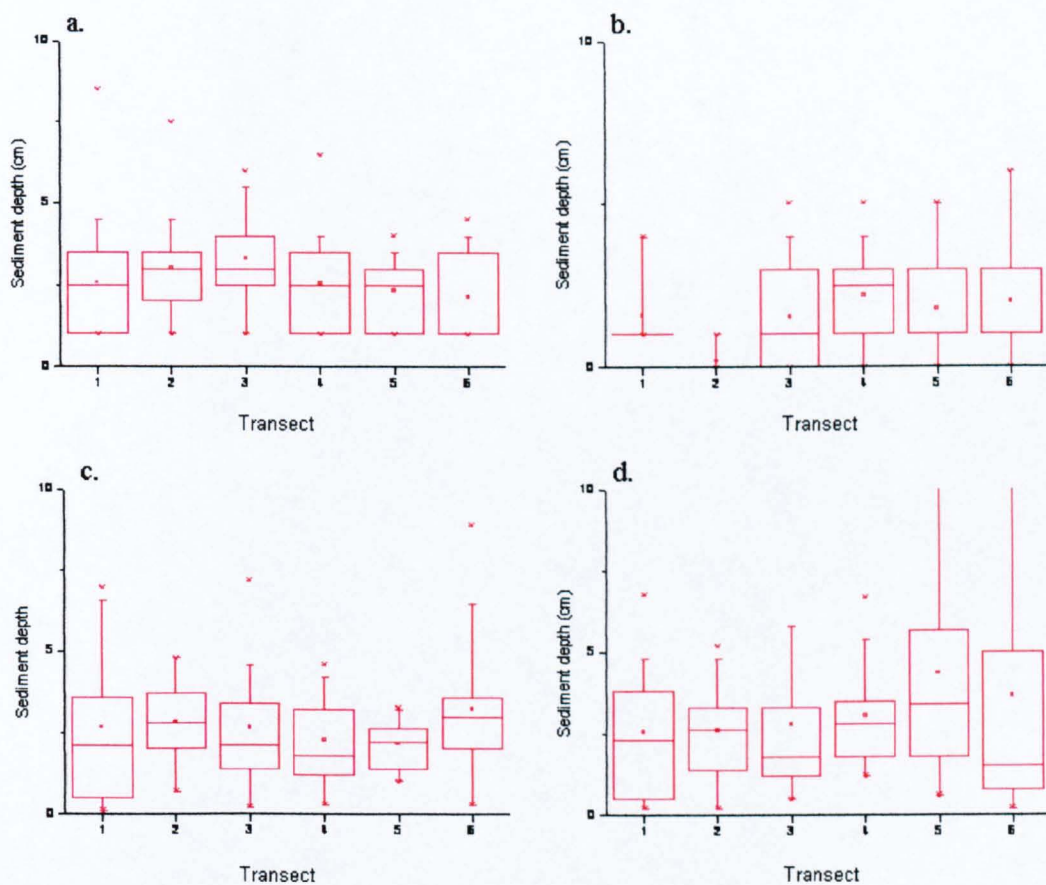


Figure 5.30 Transect box plots of sediment depth at Crockways re-executed for Transects 1 – 6 only: a). March; b). July; c). August and d). November 2004.

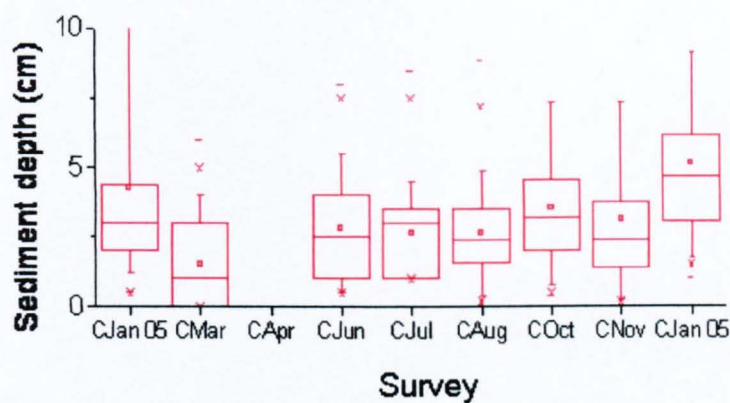


Figure 5.31 Agglomerative box plots of sediment depth at Hydeclift re-executed for Transects 1 – 8 only.

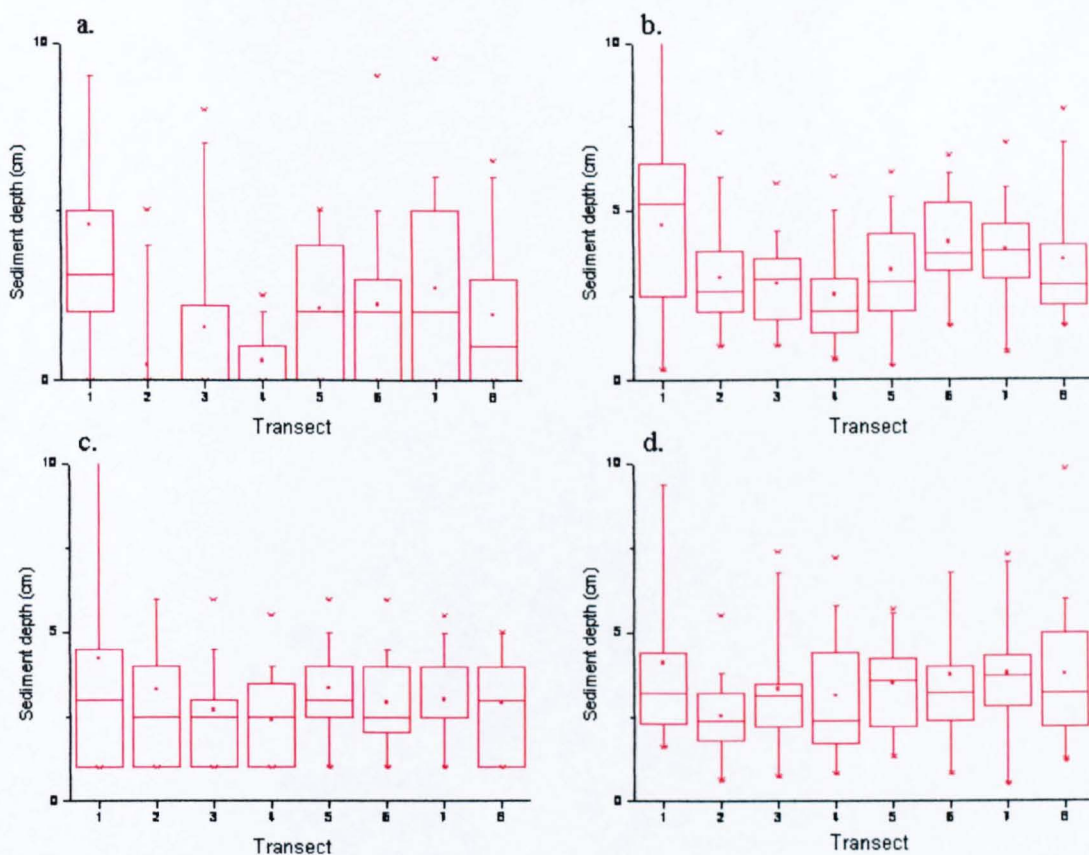


Figure 5.32 Transect box plots of sediment depth at Hydeclift re-executed for Transects 1 – 8 only: a). March; b). July; c). August and d). November 2004.

the highest sediment depth in all four surveys and this is due to its marginal location beneath an eroding bank, whereas the other T2 – T4 and T5 – T8 are differentiated by vegetation cover. This vegetation difference is also matched by a change from high velocity in the un-vegetated areas to low velocity in the highly vegetated areas and it is this transition that explains the differing sediment levels. It is unclear, however, whether sedimentation is increased in the vegetated areas, or whether erosion is increased in the un-vegetated areas. It seems likely that both processes contribute to the observed sediment pattern.

5.6 Cluster analysis

Classification of river features based on physical parameters is a popular area of research in the field of eco-hydraulics. Clifford *et al.* (2002) and Emery *et al.* (2003) explored quantitative means of generalising flow behaviour and attempted to characterise habitat units, and to assess their coherence, based on physical delimiters such as depth, water velocity and sediment. These studies centred on the relationship between classified flow types and channel bedforms and also investigated the differences in spatial location and coherence occurring as a consequence of changing flow stage. Newson *et al.* (1998) and Clifford *et al.* (in press) investigated possible linkages between physical classifications and biological classifications. Here Cluster Analysis is used to identify patches of similar velocities in the grid-reaches and attempts to investigate the changes imposed on the winter morphological velocity signal by plant growth in summer. Cluster analysis was used by Emery (2003) and Emery *et al.* (2003) and, most recently, by Gurnell *et al.* (2006) who investigated reach scale interactions between aquatic plants and physical habitat on the River

Frome (covering the Crockways and Frampton PT reaches). Gurnell *et al.* (2006) found that macrophyte growth can significantly alter summer flow patterns. Velocity was reduced in vegetated areas of the channel and was increased in 'threads' of water flow which develop between stands of vegetation and along the channel banks. They argue that these velocity changes have important implications for sediment depth and calibre and in-channel morphology. Comparisons between the present study and that undertaken by Gurnell *et al.* (2006) are made in the discussion section at the end of the chapter.

5.6.1 Introduction to cluster analysis: algorithms and similarity measures

Clustering is a classification method that assigns observations to groups, or 'clusters', such that observations within each group are as homogeneous as possible, while at the same time are as distinct as possible from the observations in other groups (Davis, 2002). The similarity of objects is based on a series of 'characteristics' for each 'observation'. In this study the observations refer to the grid squares at which measurements of physical variables were made, and the characteristics refer to the variables measured in the grid: vegetation cover; water depth; sediment depth; and water velocity in three dimensions (V_x , V_y and V_z). The clustering procedure may be based on several different characteristics of an observation at one point in time or, as used in this analysis, may be based on one or more characteristics of an observation as it changes over time.

There are many ways of grouping observations, and there are four general types of clustering procedures: partitioning methods; arbitrary origin methods; mutual

similarity procedures; and hierarchical clustering (Davis, 2002). The hierarchical clustering procedure is generally most widely used, and was chosen for use in this study. In hierarchical clustering, each observation begins as a separate cluster; the most similar clusters are joined first to form larger combined clusters and this process repeats until only one cluster remains. This final cluster contains all the original observations. Hierarchical clustering procedures require a method of calculating the similarity of pairs of observations (a similarity index) and a method for deciding which clusters are to be joined, and at which stage, in the merging process (a clustering algorithm). There are many methods available, and different merits for each method, but there is no clear answer as to which algorithm returns the 'best' results, and this introduces subjectivity into the procedure. A description of different similarity indices may be found in Davis (2002), while an evaluation of several clustering algorithms (with regard to a water velocity dataset) is presented by Emery (2003). Based on Emery's analysis, Ward's method has been chosen for this analysis along with 'Euclidean distance' as a similarity measure (see Davis (2002) for description). This is the same method used by Gurnell *et al.* (2006), and facilitates comparison between their results and those of the present study. Griffith and Arnheim state that Ward's algorithm produces "the most appealing overall results in terms of cluster size, shape density and internal homogeneity" (Griffith and Arnheim, 1997, p. 220).

The progressive merging, or 'clustering', of observations may be displayed in a dendrogram: a tree-like diagram, which contains a list of all the initial observations as a base and shows the connections made between observations during each iteration of the merging process. The most appropriate number of clusters to

describe each data set will differ depending on the natural structure of the data, and must be chosen by the user after visual inspection of the dendrogram. In this analysis, spatial mapping of several different cluster scenarios was also used to help choose the most appropriate number of clusters to represent the data.

5.6.3 Cluster analysis procedure

Cluster analysis was performed using SPSS (Statistical Package for Social Scientists) version 13.0. Missing values were removed prior to analysis and the data were arranged in columns, separated by survey. Values were standardised as z-scores and a dendrogram and agglomeration schedule were returned with each analysis. Membership data for all clustering solutions ranging from 2 – 10 groups were also returned. The dendrograms and agglomeration schedules were used to choose the most appropriate number of clusters in each analysis and the membership list for this solution was used to map the data and to assign each velocity value to its class.

Missing data in several of the Crockways surveys presented problems for the cluster procedure as cells can only be included in an analysis if a value is available for the cell in each survey. The Crockways data had to be analysed in two separate procedures, comprising: (i) all nine surveys with a spatial restriction of CS1 – CS10; and (ii) all cross-sections, CS1 – CS20, but for five surveys only: March, June, July, August and November 2004. The October survey at Crockways was not considered in the second analysis due to the change in discharge conditions between the first and last 10 cross-sections in the survey (see Figure 5.2), which may have spuriously influenced results. The Hydeclift site required only one analysis procedure and all

cross-sections in the nine surveys were considered at once. Similarly, singular cells that do not have a velocity reading attached in each survey cannot be included in the analysis. This is most problematic at the non-vertical left bank at Crockways, where falling water levels in summer meant that either these cells ceased to be submerged or the water was too shallow to permit measurement. Changing water levels and bank lines may have had an influence on the analysis. Though the cells retain the same spatial location, they may no longer have the same relation to bed and bank skin resistance and this may produce seasonal results that are unrelated to vegetation effects.

5.6.4 Choice of variables

Several variables, and combinations of variables, were considered for use in cluster analysis, including: (i) stream-wise velocity (V_x); (ii) V_x and water depth; (iii) sediment depth; and (iv) several combinations of V_x , cross-stream velocity (V_y), vertical velocity (V_z) and the standard deviation of V_x .

The V_x , V_y , V_z and standard deviation of V_x were considered as indicators of changing diversity in the flow field and also as measures of sedimentation: V_z describes the nature and strength of depositional and resuspension velocities; while the standard deviation of V_x provides a measure of turbulence, which is also linked to sediment transport. Preliminary descriptive plots of the V_y , V_z and standard deviation of V_x data for each survey revealed that summer and winter differences do exist, switching from incoherent spatial patterns in winter to greater underlying structure in summer, tentatively linked to spatial patterns of vegetation growth and it

was thought this transition might be revealed in the cluster analysis. However, these variables, whether considered singularly or in combination, did not return meaningful results in the cluster analysis, perhaps because the extremely complex and unstructured situation experienced in winter was too complicated to allow for simple clustering of the data.

Sediment depth data were also considered for clustering, but the cluster analysis returned a strong division into only two clusters. When mapped, these corresponded to a division between marginal and central areas of the channel and this suggests that the strong marginal/central division had obscured smaller scale variation. Bank sediments were shown to decrease in the marginal areas in summer while the central areas increased slightly and this may indicate a redistribution of sediment from the channel margin to the centre of the channel. However, due to problems in the collection of sediment data at the channel margins, this seasonal pattern may be an artefact of changes in sampling area (See Section 5.5.4).

The combination of V_x and water depth variables for clustering returned meaningful clusters at each site, but these were more strongly related to water depth than to velocity and the three clusters divided the channel into three sections of shallow, intermediate and deep water. This produces velocity patterns of different absolute velocity but with the same seasonal pattern, similar to that of the reach-averaged velocity patterns. Water depth was the dominant variable in the clustering process, despite standardisation of both variables using z-scores, and this served to obscure any smaller scale spatial and temporal changes in velocity.

Stream-wise velocity (V_x) was chosen for use in the clustering procedures, as the variable that provided most information as to the effects of vegetation on channel conditions and that provided clear interpretation and meaning. V_x values were re-scaled as z-scores from 0 - 1 so as to minimise discharge influence. The clustered V_x data were related to vegetation cover patterns in each reach to help explain the temporal and spatial trends observed.

5.7 Crockways: nine surveys CS1 – CS10

The V_x dendrogram for the first Crockways analysis suggested an optimum division of the data into four clusters, named C1 to C4 based on their order of formation during the clustering process (see Appendix C). Figure 5.33 illustrates the changing re-scaled stream-wise velocity for each cluster in each survey at Crockways, while Figure 5.34 shows the spatial location of cluster cells within the grid. Table 5.1 summarises the velocity characteristics of all the clusters in each of the three analyses. The average re-scaled velocity for the whole reach is also shown as a reference to help interpret the dominance of individual cluster signals. The velocity plots indicate two primary means of cluster partition: (i) a division into low and high velocity clusters, which broadly separate the marginal and central areas of the channel; and (ii) a distinction in seasonal trend within the low and high velocity classes.

5.7.1 Cluster characteristics and seasonal trends

The division into high and low velocity clusters was judged by the initial velocities in January and March, when vegetation influence was minimal. Figure 5.33 and Figure 5.34 indicate that Cluster C1 and Cluster C4 are ‘low-velocity, marginal’ groupings; Cluster C1 predominates at the vertical left bank, whereas Cluster C4 cells are located exclusively at the more gently sloping right bank. The marginal location and low water depth explains the low average velocity in both clusters. Cluster C2 and C3 are more centrally located and may be considered as ‘high velocity, mid-channel’ groupings; Cluster C2 is found mostly to the left of the channel, while Cluster C3 predominates in the centre and right of the channel. The higher water depth and central location are responsible for the high winter velocities in these clusters

Clusters C1 and C4 and Clusters C2 and C3 may exhibit very similar initial average velocities but the seasonal trend that follows is very different in each, and may be related to vegetation cover. Figure 5.35 shows a vegetation cover map for Crockways, at maximum biomass, in August 2004; while the stacked histogram in Figure 5.36 quantifies spatial correlations by displaying the percentage incidence of each cluster in each vegetation class for the August survey. The grid-plots and stacked histogram show a good spatial correlation between vegetation category and cluster number: Clusters C1, C3 and C4 are generally found in high-vegetation cells (vegetation category 4 or 5) while Cluster C2 is predominant in cells of low vegetation cover (vegetation categories 0 – 3). The high vegetation cover quoted for Cluster C1, predominantly relates to trailing riparian vegetation and this has an

Figure 5.33

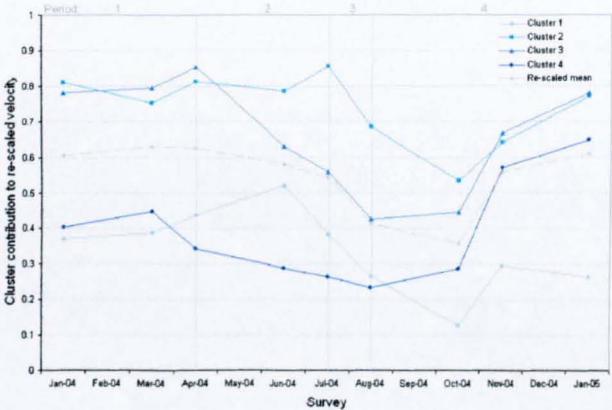


Figure 5.36

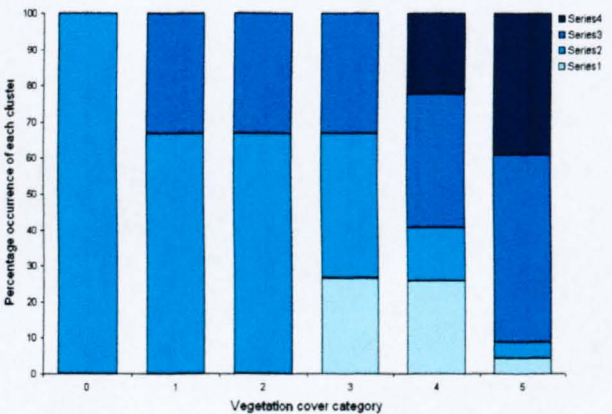


Figure 5.34

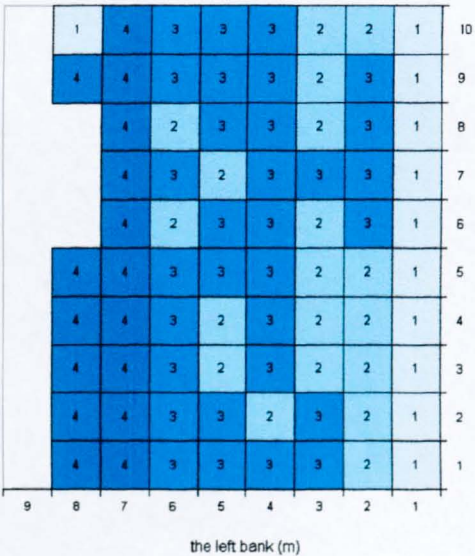


Figure 5.35

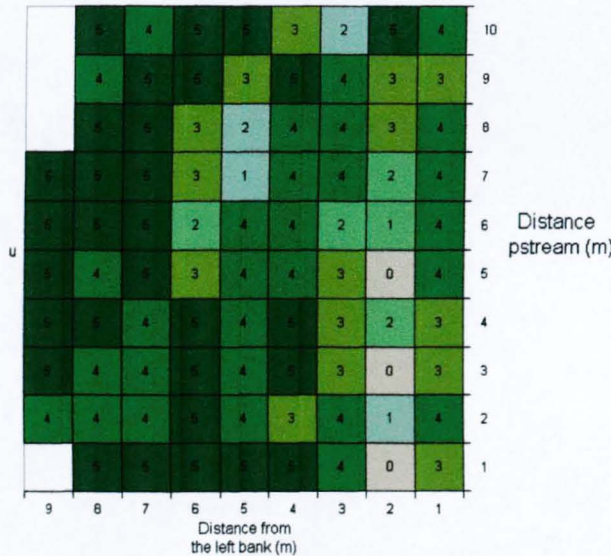


Figure 5.33 The changing re-scaled V_x for each cluster in each survey for the Crockways nine survey analysis (CS1 – CS10).

Figure 5.34 Spatial map of the Crockways grid showing the location of V_x cluster cells within the Crockways grid.

Figure 5.35 Vegetation cover map for Crockways, at maximum biomass, in August 2004.

Figure 5.36 Stacked histogram displaying the percentage incidence of each cluster in each vegetation class for August at Crockways

Table 5.1 Description of velocity characteristics and percentage coverage of the channel for each of the four clusters obtained in the three separate cluster analysis procedures.

Analysis	Cluster	Cluster characteristics	Area
Crockways CS1 – CS10	C1	Low velocity, marginal location, mainly at left bank. Increasing velocity March - June, decreasing June - October. Predominantly low vegetation cover.	14.3 %
	C2	High velocity, mid-channel location. Increasing velocity March - July, decreasing July - October. Predominantly low vegetation cover.	24.7 %
	C3	High velocity, mid-channel location. Dramatically reduced velocity April - October. High vegetation cover, submergent species.	40.3 %
	C4	Low velocity, marginal location, mostly at right bank. Stable reduced velocity March - October. High vegetation cover, emergent species.	20.8 %
Crockways CS1 – CS20	C1	Low velocity, marginal location, predominantly at left bank. Velocity Increased March - June, declining June - August. Predominantly low vegetation cover.	6.6 %
	C2	High velocity, mid-channel location, predominantly to left of channel. Increase in velocity March - July, decrease in velocity July - August. Predominantly low vegetation cover.	22.5 %
	C3	High velocity, mid-channel location. Dramatic decrease in velocity March - August, return to high velocity in November. High vegetation cover, submergent species.	44.4 %
	C4	Low velocity, marginal location, significant presence on both left and right bank, Slight reduction in velocity March - July, returning to March level by November. High vegetation cover, emergent species.	26.4 %
Hydeclift CS1 – CS20	H1	Intermediate velocity, marginal location at both banks. Generally follows average re-scaled velocity pattern; discharge dependent. Low vegetation cover.	23.1 %
	H2	Low velocity, marginal location at both banks. Increase in velocity January 2004 - October, subsequent decrease October - January 2005. Low vegetation cover.	7.0 %
	H3	High velocity, mid-channel location at centre and left of channel. Generally follows average re-scaled velocity pattern, and is greatly discharge dependent. Low vegetation cover.	51.1 %
	H4	High velocity, mid-channel location, to right of channel. Dramatic decrease in velocity from January 2004 - August. Return to high velocity October - January 2005. High vegetation cover, submergent species.	18.8 %

impact on surface velocities only. Cluster C1, may, on this basis, be considered as a low vegetation cluster and this creates a division into two low (Clusters C1 and C2) and two high vegetation cover (Clusters C3 and C4) clusters which share similar seasonal velocity trends. Clusters C3 and C4 may be further divided into clusters dominated by emergent or submergent species. Cluster C4 cells are dominated by the marginal, emergent macrophytes *Sparganium erectum* and *Phalaris arundacea*, while Cluster C3 cells are exclusively populated by submergent species, dominated by *Ranunculus calcareous*. This distinction in growth form between clusters is evident in both the relative timing and strength of seasonal velocity changes in Figure 5.33.

The seasonal trends displayed in Figure 5.33 may be divided into four distinct periods based on changes in average reach-scale velocity and reach-scale vegetation cover:

- | | | |
|--------------------------------|---|---|
| 1. January 2004 to April 2004 | – | Stable reach-scale velocity
Minimum plant biomass |
| 2. April 2004 to July 2004 | – | Gradual reduction in reach-scale velocity
Rapid vegetation growth
First critical vegetation biomass attained |
| 3. July 2002 to August 2004 | – | Dramatic reduction in reach velocity
Closing of preferential flow channels
Second critical vegetation biomass |
| 4. August 2004 to January 2005 | – | Gradual return to stable winter velocity
Senescence and washout of plants
Return to minimum plant biomass |

5.7.2 Period 1: Stable reach-scale velocities

Period 1 is characterised by stable reach-scale velocities, resulting from minimal vegetation cover and limited plant growth. The velocity distinction between marginal and central clusters is greatest in this period, while the variability within the separate marginal and central groups is very low. The first indication of vegetation influence is evident between March and April in Cluster C4; here early growth of emergent macrophytes has led to a 25% reduction in average velocity, while at the same time velocities increased in the other three clusters.

5.7.3 Period 2: Gradual reduction in reach-scale velocity

In Period 2, the four velocity clusters are responding to rapid increases in reach-scale vegetation cover. The two clusters representing vegetated areas, C3 and C4, both experience a clear reduction in average re-scaled velocity from April through to July, and this is due to the rapidly increasing vegetation cover in these cells, which increases roughness and flow resistance. There are, however, important distinctions in the timing and magnitude of the seasonal signal in Clusters 3 and C4 and these may be linked to the dominant macrophyte species in each cluster. Cluster C4 is dominated by the emergent *Sparganium erectum*, which typically has an early and rapid start to the growth season at this site, and this may explain the early velocity reduction in this cluster from March to April. This initially rapid decrease in velocity is followed by a much smaller, sustained, rate of decrease from April to August and this fits with the *Sparganium* growth cycle and growth form: following the establishment of submerged, anchoring, plant parts, *Sparganium* continues to mature

between April and August, but much of the growth is above the water surface, while at the same time lateral extension is restricted due to the deeper water and high velocities in the central channel (CEH, 2004). The continuing, low-level, decreases in velocities observed in Cluster C4 between April and August are likely due to thickening of the existing submerged plant parts and the longitudinal consolidation of individual plant stands. Cluster C3 by contrast is dominated by the submergent *Ranunculus calcareous*. This species has a later start to the growth season, but experiences rapid growth throughout the growth season with almost all growth occurring below the water surface, facilitating large velocity reductions, as evident in Figure 5.33.

In the un-vegetated clusters, velocity generally increased throughout Period 2: velocity in Cluster C1 increases greatly from April through to June, while Cluster C2 displays stable velocities from April to June followed by a dramatic increase in velocity from June to July. Cluster C1 and 2 represent cells of low vegetation cover that form a ribbon-like 'preferential flow channel' at the left bank which stretches uninterrupted from the upstream to downstream end of the reach (Figure 5.34). A smaller, more disrupted channel may also be present to the downstream right of the reach, distinguished by several cells from Cluster C2. Colonisation of these flow channels is restricted by strong shear stresses and unfavourable coarse substrates (Sand-Jensen and Mebus, 1996) and the high velocity channels act to offset decreases in velocity in the high vegetation cells of Cluster C3 and C4. The existence of similar flow channels in vegetated rivers have been described by Dawson and Robinson (1984), Machata-Weiniger and Janauer (1991), Sand-Jensen and Mebus (1996) and Gurnell *et al.* (2006).

Despite there being an equal number of increasing and decreasing velocity clusters, reach-scale velocity is shown to decline between June and July, indicating that decreases in velocity in highly vegetated areas are only partly offset by increases in less vegetated areas. This is due to differences in the relative significance of each cluster. The contribution of each cluster towards the reach-scale average is determined by: (i) the size of each cluster; and (ii) the degree of seasonal change in each cluster. The 'size' of a cluster relates to the spatial coverage of the cluster cells, and clusters that cover a large number of grid cells will have correspondingly more influence than clusters with only a small cell membership. Size is likely to be the dominant factor in any reach, but the dominance of the largest cluster may be strengthened or weakened by differences in the degree of seasonal velocity change between clusters. The relative influence of each cluster in determining average reach velocity can be examined by using the average velocity of each cluster in January as a baseline and measuring the change in each subsequent survey against this. Calculations of velocity change, weighted by cluster size, for each cluster and their contribution to average velocity are displayed in Table 5.2

In June, the average 0.15 increase in re-scaled velocity experienced in Cluster C1 was equal in strength to the average 0.15 decrease in velocity for Cluster C3 and higher than the 0.12 average decrease in C4, but the number of Cluster C2 (stable velocity), C3 and C4 cells (decreasing velocity) greatly outnumbered the Cluster C1 cells causing a decline in reach velocity and a increase in water depth (see Table 5.2). Critical biomass is first achieved between April and June when vegetated cells outnumber un-vegetated cells. These dates accord well with the estimated establishment of critical biomass in late April at the Frampton PT reach in Chapter 4.

Table 5.2 Table showing the contribution of each cluster to the seasonal change in reach scale average velocity (Vx), with reference to the winter baseline of January 2004 for the Crockways nine survey analysis (CS1- CS10) for June, July, August and October.

Survey	Cluster name	Cluster as % area of grid	Vx Change from January	Contribution to Vx change (Magnitude*Area)	Total change in reach from previous survey (re-scaled Vx)
June	Cluster C1	15.58	0.150	0.023	0.010
	Cluster C2	25.97	-0.026	-0.007	0.001
	Cluster C3	38.96	-0.151	-0.059	0.029
	Cluster C4	19.48	-0.116	-0.023	-0.012
	Totals	100	-	-0.065	-0.0923
July	Cluster C1	15.58	0.012	0.002	-0.021
	Cluster C2	25.97	0.046	0.012	0.019
	Cluster C3	38.96	-0.223	-0.087	-0.028
	Cluster C4	19.48	-0.140	-0.027	-0.005
	Totals	100	-	-0.100	-0.035
August	Cluster C1	15.58	-0.106	-0.016	-0.018
	Cluster C2	25.97	-0.126	-0.033	-0.045
	Cluster C3	38.96	-0.355	-0.138	-0.051
	Cluster C4	19.48	-0.172	-0.033	-0.006
	Totals	100	-	-0.221	-0.121
October	Cluster C1	15.58	-0.244	-0.038	-0.022
	Cluster C2	25.97	-0.277	-0.072	-0.039
	Cluster C3	38.96	-0.336	-0.131	0.007
	Cluster C4	19.48	-0.119	-0.023	0.010
	Totals	100	-	-0.264	0.043

However, the agglomerative analysis in Section 5.3.2 revealed that increases in water depth are not constant following the attainment of the first critical biomass. Instead they vary during the summer season becoming increasingly greater from June through to October. In July, two clusters (C1 and C2) show an increase in velocity relative to January, the marginal Cluster C1 shows an increase of 0.12 (a reduction from June), while the central cluster has greatly increased from January (and from June) by 0.046. The velocity increases in these clusters are, however, much less than the decreases in velocity experienced in the vegetated clusters C3 and C4 where velocity declined by - 0.223 and - 0.140 respectively. The cluster contributions in Table 5.1 show that the vegetated clusters have a greater influence because of size dominance *and* because decreases in velocity inside vegetated cells were significantly greater than the increases experienced in un-vegetated cells. This has led to a further decrease in velocity and an increase in water depth, relative to June. However, the velocity decrease in vegetated cells has been significantly offset by the increased velocity in the un-vegetated cluster C2, and the mean decrease between June and July is not as great as the difference between April and June.

5.7.4 Period 3: Dramatic reduction in reach velocity

Period 3 includes the large and rapid fall in reach-scale velocity between July and August (Figure 5.33). In August, all four clusters demonstrate a decline in velocity from July levels and this corresponds well to the dramatic fall in reach-scale velocity between these surveys. It may be ventured that *two* distinct and successive critical biomass levels exist. The first is attained by June in Period 2, when cells of high vegetation cover, and reduced velocity, outnumber compensatory low vegetation

cover cells of increased velocity. The second critical biomass is attained in August and describes the point when vegetation cover increases to a level such that velocity is reduced in all areas, whether densely or sparsely vegetated. This second critical biomass may be a result of 'closing' of preferential flow channels described in Period 2. A visual comparison of vegetation cover maps at Crockways in July and August reveals isolated cells of high vegetation cover present in August (but not in July) which may act as vegetation 'dams', breaking up the ribbon-like flow channels and reducing velocity in the less vegetated cells downstream and perhaps facilitating the establishment of vegetation in these cells. In Table 5.2 all four clusters are shown to decrease in velocity between the July and August surveys, though the decrease relative to January is still greatest in the vegetated clusters. This reduction in un-vegetated cells means that the offsetting effect from the un-vegetated preferential flow channels is reduced and this leads to a dramatic reduction in average reach-scale velocity, the largest decrease observed between any two surveys.

This second critical threshold could be more related to a pattern of vegetation growth and not just a result of increased biomass: if plants are arranged in such a way that they join across the channel from bank to bank, then even at low biomass this second critical event may be achieved. However, aquatic plants naturally grow in a staggered pattern with gaps between plants maintained by the accelerated velocities in the preferential flow channels. The second critical threshold is likely to be biomass driven because colonisation of un-vegetated high velocity areas is not possible without a reduction in overall reach velocity, and this is only achieved under very high biomass. The closing of the preferential flow channels dramatically

decreases velocity in the reach and could possibly lead to complete coverage of the channel if not for the natural autumnal decline of the plants.

5.7.5 Period 4: Gradual return to stable winter velocities

Period 4 sees a return to the stable velocity conditions experienced in Period 1 and the reverse of patterns observed in Period 2. The trends in the clusters have changed once more and the ‘vegetated’ cells (Clusters C3 and C4) now show an *increase* in velocity, relative to August, while the ‘unvegetated’ cells in the blocked flow channels show a continuing decline in velocity as immature vegetation stands increase in biomass. Overall water velocity continues to decline in October, relative to January, despite an increase in velocity in the larger clusters and this suggests that cluster size is no longer the dominant influence on reach-averaged velocity. Table 5.2 shows that the velocity change in the smaller Clusters C1 and C2 between August and October is much stronger than the velocity increase in the larger clusters C3 and C4 and this is able to override the size dominance. The rate of velocity decline from August to October is much less than July to August, however, and this shows the continuing influence of the larger clusters. Of the nine surveys included in this analysis, the reduction in velocity, relative to January, is greatest in October, just before senescence of the plants.

The reversal in behaviour between the ‘vegetated’ and ‘un-vegetated’ clusters may be linked to differences in the maturity of the plants in each cluster. In Clusters C3 and C4 vegetation stands are large and long-established which makes them more susceptible to washout during high discharges than the smaller plants in Cluster C1

and 2. Fine sediment accumulation in Clusters C3 and C4 may also allow vegetation in these clusters to be preferentially removed during high discharge (Haslam, 1978), while less mature plants, with firmer anchorage sites, i.e. less accumulated fine sediment (Haslam, 1978), and more flexible stems in Clusters C1 and C2 are able to resist washout. After October, the velocities in all clusters increase, heralding a return to the higher, stable, velocities of Period 1.

5.8 Crockways: five surveys CS1 – CS20

In the second Crockways analysis, seasonal detail is reduced while spatial coverage is increased and the analysis is mainly used to elucidate the explanations of spatial patterning developed in Section 5.7. A four-cluster division was again chosen as the optimum representation of the data and this promotes continuity between the first and second analyses. The velocity characteristics of these clusters are illustrated in Figure 5.37 and summarised in Table 5.1. Spatial maps of cluster cell locations and vegetation cover at maximum biomass in August are provided in Figure 5.38 and 5.39; while Figure 5.40 displays a stacked histogram which shows the percentage incidence of each cluster in each vegetation class in August.

The four clusters exhibit very similar characteristics to clusters obtained in the first analysis: there is a similar distinction between high and low velocity clusters and the seasonal trend though reduced in detail, generally match the summer patterns from the first analysis (Table 5.1). The clusters are not strictly the same for the two analyses but they are sufficiently similar to be assigned the same names to aid interpretation. The greatest difference between the nine-survey and 5-survey

Figure 5.37

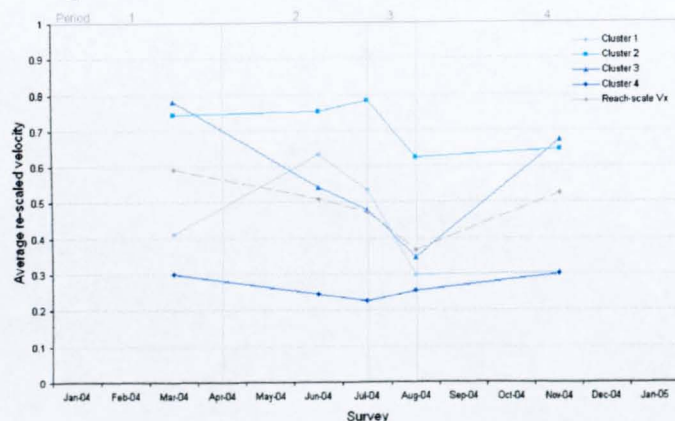


Figure 5.40

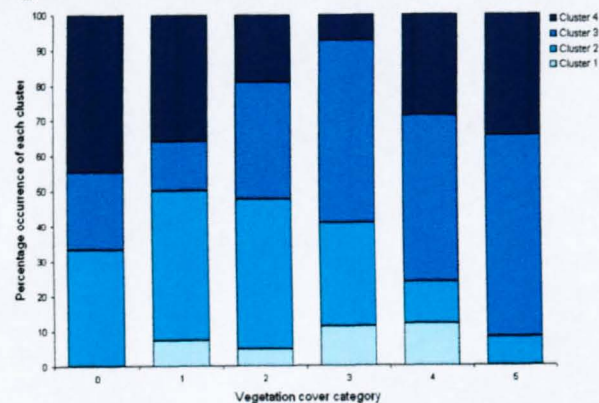


Figure 5.38

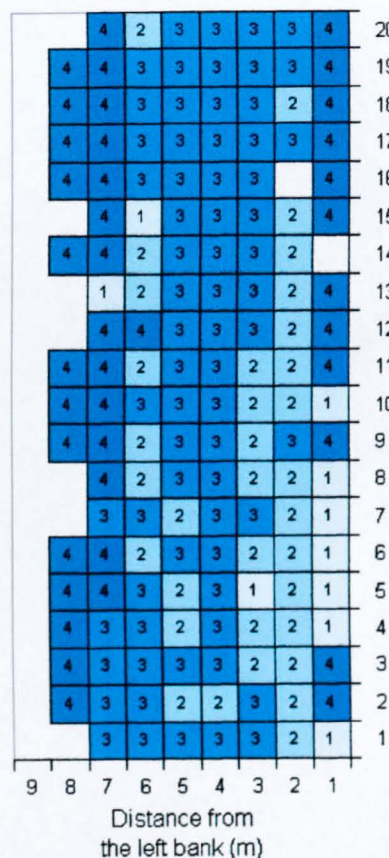


Figure 5.39

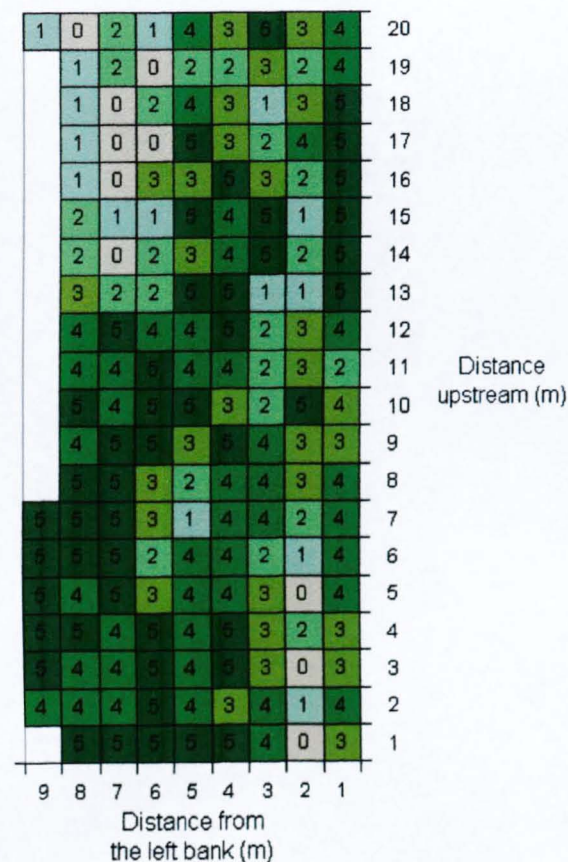


Figure 5.37 The changing re-scaled Vx for each cluster in each survey for the Crockways five survey analysis (CS1 – CS20).

Figure 5.38 Spatial map of the Crockways grid showing the location of cluster cells within the Crockways grid.

Figure 5.39 Vegetation cover map for Crockways, at maximum biomass, in August 2004.

Figure 5.40 Stacked histogram displaying the percentage incidence of each cluster in each vegetation class for August at Crockways.

analyses is in Cluster C4; the absolute velocity is reduced from the first analysis and the seasonal pattern has changed slightly. There is now a slight increase in re-scaled velocity between July and August, whereas in the first analysis there was a decrease during this period. Another obvious change between analyses is in the magnitude of seasonal change in Clusters C1 and C3. In the first analysis, velocity values in these clusters became increasingly convergent, but Cluster C3 always remained above Cluster C1. In the second analysis the lines for these cluster cross in June and July and Cluster C1 attains temporarily higher velocity than Cluster C3, due to a marked increase in the velocity of the Cluster C1 signal. These two changes may be due to the inclusion of new cells, or to the loss of old cells to other clusters, which alters the average velocity characteristics of Cluster C1.

Changes in absolute velocities and seasonal patterns between analyses may be related to changes in spatial location and vegetation abundance in the cells of each cluster. The clusters in the second analysis are generally found in the same spatial locations as in the first, but some distinctions can be made. At a crude scale, the distinction between marginal, low velocity and central, high velocity clusters has remained similar between the two analyses, with an almost exact 2:1 ratio (by area) of mid-channel to marginal cells in each analysis (see Table 5.1). The twofold distinction between marginal and central cells accurately reflects the channel morphology but masks some significant seasonal changes.

5.8.1 The margins

In the marginal areas, the dominance of Cluster C4 has increased, from 21 % to 26 %, while Cluster C1 cells have declined from 14 % spatial coverage to 6 % (Table 5.1 and Figure 5.38). The decline in the areal extent of Cluster C1 does not denote a change in cell designation from Cluster C1 to other clusters, but mainly reflects the dominance of Cluster C4 cells in the upstream portion of the grid. Cluster C1 cells are confined to the downstream left bank and this generally corresponds to deeper marginal areas and to where the bank is undercut. This morphological niche may be actively created by the vegetation: the high-velocity ‘flow-channels’ that travel close to the left bank in summer may cause erosion of the bed and bank, whereas the right bank is protected by a buffer strip of emergent vegetation (Gurnell *et al.* 2006).

Cluster C4 predominates in all other marginal areas, and it is interesting to note that cells from the pool area are included in Cluster C4 and not set apart as a separate cluster. The addition of pool-cells to Cluster C4 may explain both the relative reduction in absolute velocity and the slight change in seasonal pattern observed between the first and second analyses. The effects of the pool cells are also evident in changes in vegetation abundance: in the first analysis this cluster was only present in high vegetation categories, while in the second analysis the stacked histogram shows that Cluster C2 is equally prevalent in both high and low vegetation classes (Figure 5.36 and Figure 5.40). This new binary distribution is related to the inclusion of low-vegetation pool cells in this cluster. Given their low vegetation cover, pool cells may have been expected to have more similarities with Cluster C1; low-velocity, low-vegetation cells that experience a compensatory increase in velocity in

summer. The association of the pool cells with Cluster C4 however, may suggest a depth-limitation to vegetation effects, with the pool cells not preferentially increasing or decreasing in velocity but adopting a velocity signal similar in pattern to the reach-scale average. At the reach-scale, velocity is dominated by signals from Cluster C3 and this should be reflected in the velocity patterns in the pool, meaning pool cells will be more similar to Cluster C4 than to Cluster C1.

To check the plausibility of this hypothesis, the pool cells need to be separated from the marginal cells in Cluster C4. However, when 5 and 6-cluster solutions were mapped, they did not provide a more accurate designation of the pool cells and it must be assumed that the consistently low absolute V_x in both the marginal and pool cells means they are difficult to separate in the clustering process despite differences in seasonal pattern. The area of the pool was delineated in Section 5.4.2 and Section 5.5.2 and this information may be used to manually separate the pool cells from other cells in Cluster C4. Figure 5.41 shows the velocity characteristics for: (i) all Cluster C4 cells; (ii) pool cells only; and (iii) all remaining ‘non-pool’ cells. This reveals that the non-pool cells have a velocity pattern more similar to Cluster C4 in the first analysis, while the pool cells are wholly responsible for the changes observed in Cluster C4: the pool cells cause both the decline in absolute winter velocities between analysis 1 and 2 and also the increase in velocity between July and August. The pool cells are the only cells where velocity is observed to increase between July and August, and this suggests that the deep pool areas have a high ‘immunity’ to vegetation effects in Period 3, but are ‘activated’ in Period 4, following the closure of the flow channels in Cluster C2. The pool is an area of deep, slow water and acts as a diversionary feature: most water approaching the pool accelerates around it rather

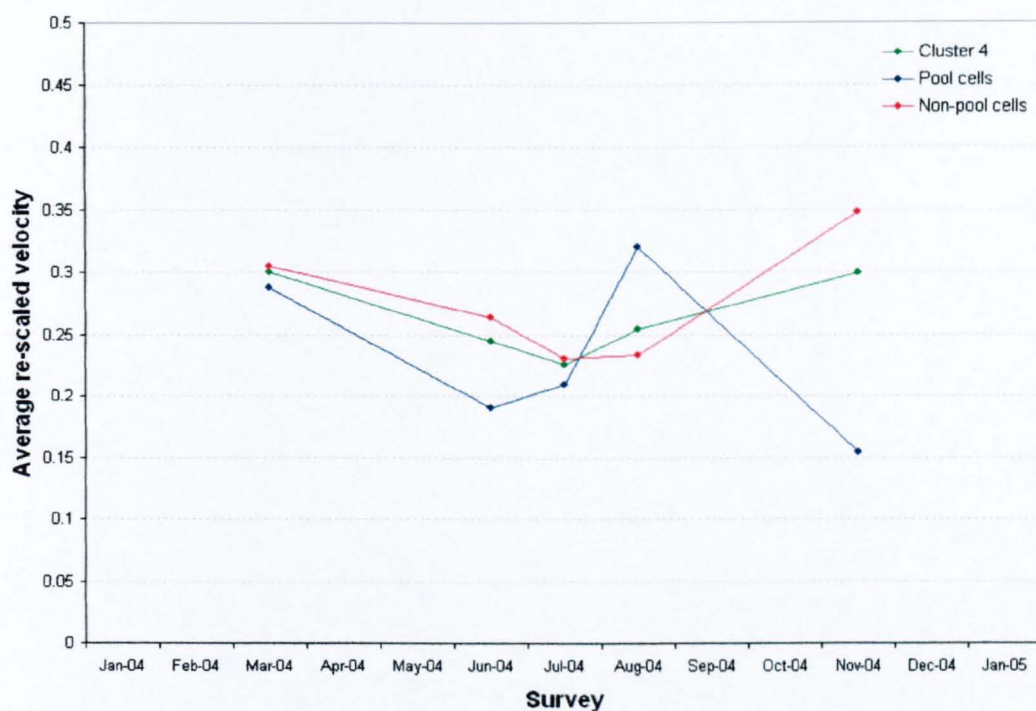


Figure 5.41 Comparison of pool and non-pool cells against the average velocity behaviour in Cluster 4 in the Crockways five survey analysis (CS10 – CS20). The pool cells experience a increase in velocity from June August, while the non-pool cells show a decrease in velocity over the same period.

than travelling through. However, when the shallow preferential flow channels are closed, the low vegetation cover in the pool means that it becomes a preferential flow route and is 'activated' at high channel biomass.

5.8.2 Mid-channel

In the mid-channel clusters, Clusters C2 and C3, spatial coverage has remained broadly similar to that observed in the first analysis. Cluster C2 has declined slightly from 25 % spatial coverage to 23 %, while Cluster C3 has increased its dominance to 45 % spatial coverage from 40 % in the first analysis. These changes are small when considered singularly but together constitute a combined relative increase in the coverage of the high biomass Cluster C3 by 7 %. The spatial cluster map in Figure 5.38 confirms the existence of two separate flow channels in the reach formed by cells from Clusters C1 and C3. The largest and most coherent of these runs close to the left bank, and is virtually unbroken from the upstream to downstream end of the grid. A smaller and less well connected flow channel is also evident to the right of the channel, and this possibly forms in the transition zone between emergent and submergent macrophyte populations. Both flow channels help to maintain efficient throughput of river discharge in the Period 2 surveys, June to July, but as in the first analysis these flow channels cease to function when they are interrupted by vegetation dams. The two flow channels seem to converge at the downstream left of the channel, possibly due to a large 'dam' of emergent vegetation evident to the downstream right (Figure 5.39). For Clusters C2 and C3 the vegetation correlation remains largely the same as in the first analysis: Cluster C2 is still more prevalent in the low vegetation categories, though its dominance in categories 0 –3 have been

offset by increases in Cluster C4, and Cluster C3 is still dominant in the high vegetation categories.

5.9 Hydeclift: nine surveys CS1 – CS20

At Hydeclift, the clustering dendrogram again suggested that four clusters best represent the seasonal variation in the velocity data. However, the characteristics of these four clusters differ greatly from those in the Crockways analyses and are named H1 – H4. Figure 5.42 illustrates the changing velocity characteristics of each cluster in each survey, and Figure 5.43 shows the spatial location of the cluster cells and both demonstrate that the clusters may be logically divided by distinctions in both absolute velocity and seasonal pattern. Absolute velocity distinctions are again judged by initial, winter, velocities and three clustering velocities are evident: Cluster H1 represents cells of moderate velocity, found in relatively deep marginal areas, while Cluster H2 represents shallow, low-velocity marginal cells and Clusters H3 and H4 are indicative of slightly deeper and faster flowing mid-channel areas. The moderate velocity cells observed in Cluster H1 are greater in number than the low velocity cells and this is indicative of the uniformity of the Hydeclift reach, where marginal and mid-channel areas are less distinct than at Crockways.

Differences in seasonal trend may again be linked to vegetation cover. Figure 5.44 shows the corresponding vegetation cover at Hydeclift in August 2004 and Figure 5.45 attempts to quantify any spatial correlations, by displaying the incidence of each cluster in each vegetation category. The most immediate observation is the strong correlation between Cluster H4 with high vegetation categories. The vegetation map

suggests a general agreement between the location of high vegetation areas and the spatial location of Cluster H4, e.g. both high vegetation cover cells and Cluster H4 cells predominate to the right of the channel. The stacked histogram confirms this association and illustrates a clear increase in the incidence of Cluster H4 with increasing vegetation cover. Correspondingly, the incidence of the other high-velocity, mid-channel cluster, Cluster H3, is dominant in cells of lower vegetation cover (categories 0 – 3) but is less well represented in the high vegetation cover categories. Clusters H1 and H2 do not appear to have any significant spatial correlation with any particular vegetation class but it may be that the dominance of the two larger clusters has obscured patterns in the smaller clusters.

5.9.1 Hydeclift cluster distinctions: affected and unaffected by vegetation

The four distinct time periods described for the Crockways reach are not appropriate for the Hydeclift site, because plant-induced water depth changes do not occur but are marked in Figure 5.42 for comparison. Instead the Hydeclift clusters may be divided into two groupings, clusters: i) directly; and ii) indirectly affected by vegetation growth.

Clusters H2 and H4 represent clusters affected by vegetation growth, but in different ways. Cluster H4 corresponds to high vegetation cover cells, which experience marked reductions in velocity from March to August, followed by a rapid return to initial, high-velocity levels by October. Cluster H4 is dominated by *Ranunculus calcareous* but demonstrates an earlier decrease in velocity than the corresponding

Figure 5.42

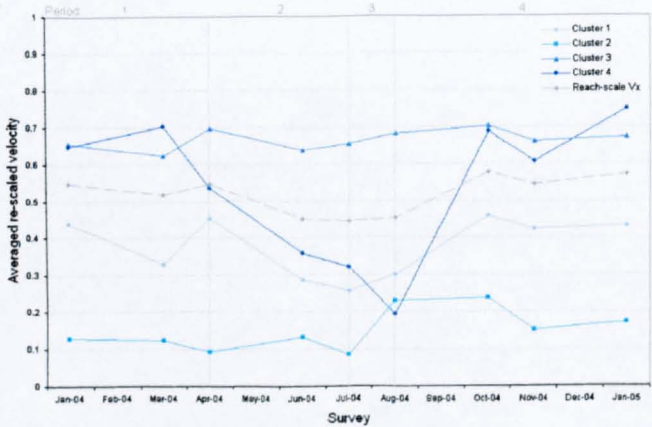


Figure 5.45

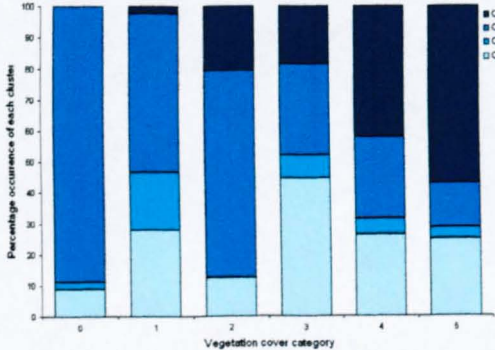


Figure 5.43

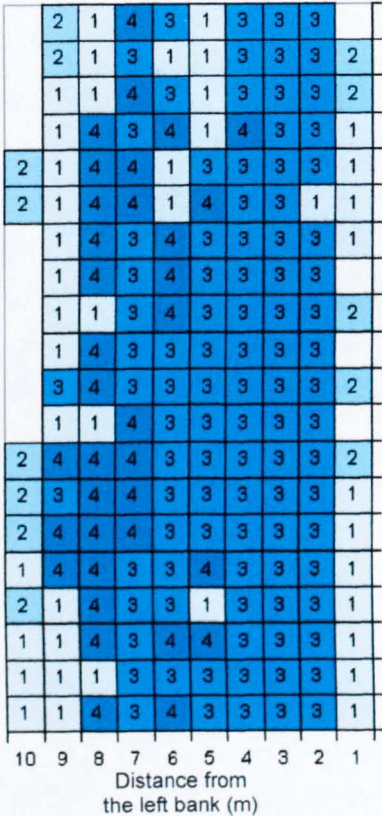


Figure 5.44

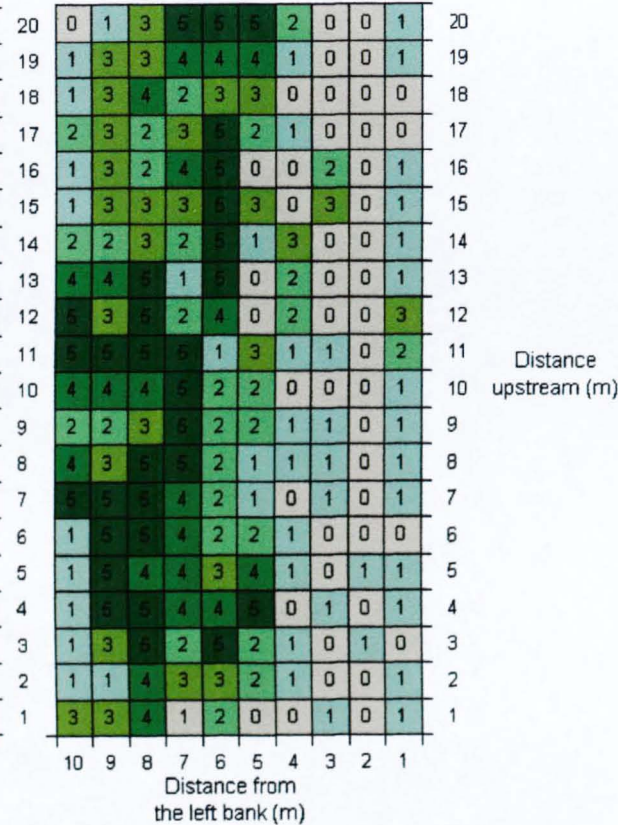


Figure 5.42 The changing re-scaled Vx for each cluster in each survey for the Hydeclift analysis (CS1 – CS20).

Figure 5.43 Spatial map of the Crockways grid showing the location of cluster cells within the Hydeclift grid.

Figure 5.44 Vegetation cover map for Hydeclift, at maximum biomass, in August 2004.

Figure 5.45 Stacked histogram displaying the percentage incidence of each cluster in each vegetation class for August at Hydeclift.

Cluster C3 in the Crockways analysis. This earlier onset to the growing season may be due to the over-wintering of dormant *Ranunculus* observed in this reach, perhaps due to the lower depths and greater light availability. These dormant remnants allow faster establishment of *Ranunculus* in spring than at Crockways. The velocity decline in Cluster H4 is very strong; the average velocity in H4 cells is the highest observed in all clusters in March, while in August H4 records the lowest velocity of all clusters. However, the spatial coverage of Cluster H4 cells is low (18.8%) and their influence on average reach-scale velocity is low. Cluster H2 exhibits a seasonal pattern that may indicate an indirect influence of vegetation growth. Cluster H2 cells generally exhibit a constant, low velocity signal from January 2004 to January 2005, but this oscillates slightly in approximate phase with discharge (see Figure 5.2). However, from July to August the velocity increases markedly, despite only a small increase in discharge perhaps indicating the development of a narrow preferential flow channel along the right bank.

Clusters H1 and H3 are the dominant clusters in the Hydeclift reach; they represent cells of moderate and high absolute velocity, and have by far the greatest spatial area, covering 23.1 % and 51.1 % of the grid area respectively. Cluster H3 displays generally stable average velocities from January 2004 through to January 2005, indicating independence from seasonal vegetation effects. Average velocity fluctuates only slightly between surveys and in approximate phase with discharge changes (see Figure 5.2), which indicate that there is a small residual discharge influence in the re-scaled data. Cluster H1 exhibits a generally similar pattern to Cluster H3. The average velocities in Cluster H1 are more variable than in Cluster H3 but generally fluctuate around a constant mean level. The marginal location of

Cluster H1 cells may mean they are more vulnerable to the effects of fluctuating discharge and this may explain the greater variability in the data.

The reduction in velocity observed in the vegetated Cluster C4 is much stronger than any of the signals in the other three clusters, however, Clusters H1, H2 and H3 together occupy 81.2% of the channel area and have an overwhelming influence on reach-scale velocities. This means that mean reach velocity is unaffected by velocity reductions within the vegetation and water depth is not influenced by seasonal vegetation growth.

5.10 Discussion

This chapter has employed four different analytical methods (both exploratory and classificatory) to examine the same data sets. The exploratory methods of analyses, both agglomerative and segregated (cross-sections and transects), were necessary pre-cursors to the cluster analysis and provided essential background information to aid interpretation of the clustering results, e.g. examination of the underlying physical structure at each site. Overall, more was revealed about in-channel patterns of vegetation growth, and vegetation interaction with physical variables, in the transect-based data when compared to either the agglomerate survey scale or the cross-sectional data. This is not surprising given: (i) the typical zoning of plants growth form across the channel; and (ii) because plant stands generally extend downstream in the direction of water flow while their lateral expansion is much more restricted (Sand-Jensen and Mebus, 1996). The transect data represent a departure from the normal survey and analysis of ecological data in rivers (for example, the

River Habitat Survey and PHABSIM both consider ecological data on a cross-sectional basis) and of recent studies such as Cotton *et al.* (2006), Gurnell *et al.* (2006), and Wharrton *et al.* (2006). Future analysis should consider the use of grid and not just cross-sectional measurements to obtain a more integrated picture of river processes.

The classification obtained through cluster analysis has definite advantages over the exploratory analyses: it is able to reconcile cross-sectional and longitudinal patterns and allows all surveys to be considered at once to separate the seasonal trends of vegetated and un-vegetated areas. However, the interpretation of the cluster groups is greatly aided by the earlier analyses, especially in terms of the influence of changing reach-scale water velocity on water depth. The cluster analysis only proved suitable for consideration of velocity, while the other analyses also considered vegetation cover, water depth and sediment depth.

The clustering method follows that of Gurnell *et al.* (2006) but the current data are more spatially and temporally detailed than that used in the previous study. Gurnell *et al.* collected their data using cross-sections, not grid-measurements, and these cross-sections were separated by an average of 6 - 11 m. Two cross-sections in their analysis may cover the same channel area as 20 cross-sections in the present study. Gurnell *et al.*'s method gives greater channel coverage but may miss the vital interconnections which occur at a very small scale between neighbouring plants. Gurnell *et al.*'s analysis also considers a smaller number of surveys (four between March and August 2004) than the present study and lacked a critical winter baseline by which to judge seasonal change. The importance of a large number of surveys

may be highlighted by a comparison of the nine-survey and five-survey analyses at Crockways in the current analyses: the overall winter to summer seasonal pattern remained very similar in both of the Crockways analyses, but the reduction in temporal detail in the five-survey analysis significantly altered the detailed interpretation of the results. For example, the omission of the winter baseline survey from January 2004 in the five-survey analysis meant that the magnitude of seasonal change in clusters could not be fully determined. This was particularly important with regard to Cluster C4 which experienced early velocity reductions in Period 1. In addition, the removal of the October survey meant that the continued velocity reduction in Cluster C1 and Cluster C2, which tempered reach-scale velocity increases in this period, was not evident. A full seasonal data set, including spring, autumn and winter surveys, and not just summer surveys, is required to fully appreciate the temporal complexity of the seasonal velocity changes occurring in a vegetated reach.

5.10.1 Site differences

The analyses in this chapter have confirmed the importance of the site-specific factors in determining the influence of vegetation in the river environment. The shaded Hydeclift site showed no evidence of plant-induced increases in water depth between winter and summer surveys, whereas the Crockways site experienced an approximate 0.16 m increase in water depth, at similar discharges, between the June survey and the August survey. The reach-scale agglomerative analysis in this chapter also provided important comparisons with the PT analyses in Chapter 4, and confirmed the existence of significant site differences. The PT and grid analyses are

mutually reinforcing. For example, the PT data averages seasonal information and this meant that it did not pick-up on shorter-term trends in velocity and water depth and may have underestimated the maximum effects of vegetation on water depth at the Crockways and Frampton PT reaches which were revealed by the grid analyses. Similarly, the averaging nature of the PT data allowed seasonal changes in sediment depth to be observed, whereas in the grid data the suspected seasonal changes were obscured by discharge effects and antecedent conditions. That the magnitude of site differences in this chapter are similar to those found in Chapter 4, indicates that riparian shading and channel morphology may have comparable effects on vegetation cover and vegetation influence. Riparian shading has been investigated in previous research (Dawson, 1979; Wright, 1982; Flynn *et al.*, 2002) and this analysis confirms that planting of riparian trees may achieve the same management control as the wholesale river rehabilitation suggested in Chapter 4 (i.e. increasing morphological diversity in the channel and incorporating deep areas that have limited light availability and will not support macrophyte growth) but at a fraction of the cost.

The comparison of the Crockways PT reach from Chapter 4 and the Crockways grid reach in this chapter also revealed that 'site' differences may be highly localised and that connectivity of water depth within a vegetated river reach is not as strong as commonly thought. The PT cross-section at Crockways is only three metres downstream of the first cross-section of the grid reach, yet it exhibits significant differences in the influence of vegetation on water depth. It was also suggested that velocity and water depth at a cross-section are more affected by the level of vegetation cover downstream of a cross-section than the level present upstream. This

limited spatial influence of vegetated areas may be very important for vegetation management: it may be that even very subtle differences in channel morphology or riparian shading may help to manage vegetation effectively.

The cluster analysis in this chapter also explored the difference in velocity influence between vegetated areas of different plant growth form (Section 5.7.3). The Crockways analyses show that cells dominated by the emergent *Sparganium erectum* have an influence on velocity earlier in the year, due to their earlier re-growth and rapid increase in biomass. This early velocity reduction was rapid between March and April but this continued at a lower, sustained level for the rest of the summer. In contrast, the cells occupied by the submergent *Ranunculus calcareous* show a later velocity reduction, first apparent between the April and June surveys. This decrease becomes stronger as the growing season progresses, and soon outpaces the *Sparganium erectum* cluster. At Crockways the emergent plants cover less of the channel area and experience a smaller reduction in velocity from winter to summer than the emergent species and, consequently, have a lesser influence on reach velocities. Emergent species, however, may have a greater impact at high discharges when more of their biomass is submerged. It is also thought that the emergent/submergent mix determines the location of the main 'preferential flow channels' which carry flow at higher velocities. The preferential flow channels tend to occur in the transition zone between emergent and submergent species where conditions are least favourable for either growth form.

5.10.2 Spatial patterning and variability within the channel

One of the main aspects of vegetation influence explored in this chapter was the spatial patterning of sediment and velocity within the reach and how this patterning changed from winter to summer. The sediment depth data proved to be highly influenced by antecedent discharge conditions, which obscured seasonal differences, and the analyses produced only limited insights into sediment patterning. The transect analyses did show that the dominant storage area in both winter and summer was at the channel margins and that it is likely that the sediment increases observed under vegetation by previous researchers (Dawson, 1978, Sand-Jensen and Mebus, 1996, Cotton *et al.*, 2006; Wharton *et al.* 2006) were due to the trapping of new sediment in the reach and not the reorganisation of existing sediments in summer from the channel margins to the centre. The transect analyses also illustrated that sediment depth in the vegetated transects at Hydeclift was significantly higher than in the un-vegetated transects. Unfortunately, it was not possible to determine whether this difference was due to increased deposition in the vegetated areas or increased erosion in the un-vegetated areas, but it is likely that both processes played a part.

It was possible to minimise discharge effects upon the velocity data to reveal the seasonal changes in velocity patterning and variability. The analyses revealed that reach-scale velocity variability was reduced in Crockways in summer, which is in opposition to the findings of previous research that generally suggests that vegetation creates more varied flow conditions and habitats for biota (Jones *et al.*, 1994). At Hydeclift, variability is greatest in summer and this reflects the fact that mean reach

velocity was not affected by the vegetation: decreases in the vegetated areas of the channel have been offset by increases in un-vegetated areas and this creates more varied velocity conditions in the channel. It would seem that velocity heterogeneity and habitat diversity increase in vegetated streams until a 'critical biomass' is reached, and after this point velocity variability is progressively reduced and may be reduced beyond that which exists under winter conditions. The analysis in Section 5.3.6 revealed that diversity of flow *direction* in the horizontal flow field increased at both sites in summer, and more so at Crockways than at Hydeclift. This suggests that diversity at high biomass sites is provided by changes in flow direction and not by changes in velocity magnitude.

The analyses also revealed the existence of ribbon like preferential flow-channels at Crockways which are the major flow paths through the reach in summer. Similar flow channels were identified by Sand-Jensen and Mebus (1996) Gurnell *et al.* (2006) and Cotton *et al.* (2006) The flow channels occurred in the transition zone between emergent and submergent vegetation where conditions are least favourable for colonisation by plant species of either growth form. It is thought that these flow channels may be blocked by encroaching vegetation at a second critical biomass in August (see Section 5.7.4), leading to a dramatic reduction in average reach-scale velocity and a marked increase in water depth. There is also evidence that after the preferential flow channels become blocked, the deeper 'pool' cells in the Crockways reach begin to show an increase in velocity. This suggests that there is a 'depth limitation' to the effects of vegetation at Crockways before the second critical biomass is reached. After the critical biomass is reached these deep areas may provide important areas of faster velocity water flow.

5.10.3 Critical biomass 1 and 2 and the timing of vegetation influence

The cluster analyses have shown that the velocity characteristics of cells in a reach are determined by the vegetation cover in their cells, but also by the vegetation cover in the reach as a whole: high vegetation cover in a section of the channel will reduce velocity in this cluster and will also increase velocity in compensatory clusters of low vegetation cover. Spatial coverage was the most influential factor at both study sites in determining whether critical biomass is attained, but the strength of seasonal change was also very important and may, at times, override the dominant influence of cluster size, e.g. in October at Crockways. Critical biomass was first achieved when un-vegetated cells were outnumbered by vegetated cells (April - June), but vegetation effects increased in July when plant stands become more consolidated and the reduction in vegetated cells was much greater than the increases in un-vegetated cells. A second critical biomass is attained in August when all clusters show a decline in velocity. The rate of velocity decline was much reduced between August and October, but, when referenced to the January baseline, velocity reduction is greatest in October.

The existence of a second critical biomass, with more extreme effects on water depth than the first critical biomass, may have important management implications. If the second critical biomass is linked to the closing of preferential flow channels, then the periodic removal of individual plants to maintain free-flowing channels may keep vegetation effects below a certain level. This would require less clearance of the channel at any one time, and would cause less ecological disturbance, but would require more regular management to ensure the flow channels remained clear. There

is also an indication that deep pool areas become more important at high biomass, they are 'activated' after the closing of shallow preferential flow channels and carry water at higher velocities. The incorporation of deep water areas in a reach could help maintain areas of fast flow in a vegetated reach.

6. MICROSCALE SEDIMENT INVESTIGATIONS

6.1 Chapter synopsis

This chapter investigates the effects of macrophytic plants on sediments and sedimentation at the microscale. The preceding chapters have demonstrated that macrophytes act to reduce water velocity and promote sedimentation, with an attendant influence on in-channel morphology, but little was known of the mechanisms of sediment capture or of the volume and size characteristics of the trapped sediment. This chapter outlines the current knowledge base regarding macrophyte-sediment interactions and critically examines the methods used in previous research. A conceptual model is developed which amalgamates previous research findings, and which serves as the basis for an experimental field method, designed to record the filtering effects of macrophytic vegetation on the sediment concentration and size composition of suspended sediment. Several key questions are examined, using a novel, experimental, methodology, which highlight gaps in current knowledge:

- What are the changes in suspended sediment concentration before flow enters and after flow exits vegetation stands? What changes occur within the vegetation? How does this compare to what happens outside the vegetation in the ambient flow stream?

- Is sediment retention within macrophytes size selective? What is the range of particle sizes retained by the plant and which sizes are not retained? Do both suspended sediment and bedload contribute to deposition within macrophytes?
- How do macrophyte-sediment interactions change with differences in vegetation characteristics, water velocity and water depth?

6.2 Macrophytes as sediment filters

Macrophyte-sediment interactions are a much more recent topic of enquiry than investigations into vegetation effects on water velocity and river stage. Initial theories as to the effects of macrophytes on sediment processes emerged from earlier velocity-centred research, often with no specific field measurement of sediment retention or sediment processes, and primarily under the premise that macrophytes increase sedimentation rates by reducing water velocity (Dawson and Robinson, 1984; Marshall and Westlake, 1990; Sand-Jensen and Mebus, 1996). Recently, and predominantly since the late-1990s, macrophyte-based research had become more sediment-specific. Several authors have attempted to quantify the retention of sediment and to identify any changes to sediment processes occurring within macrophyte beds.

The role of macrophytes in sediment transport and storage may be viewed as that of a filter (Merezhko, 1973; Marshall and Westlake, 1978; Gregg and Rose, 1982; Thornton, *et al.*, 1997; Koetsier and McArthur, 2000; Schulz *et al.*, 2003) or sieve (Carpenter and Lodge, 1986; Vermaat *et al.*, 2000; Horvarth, 2004). Like any filter,

macrophytes selectively retain in-flowing material, and may change the concentration and particle size distribution of out-flowing transported sediment (Koetsier and McArthur, 2000). Authors have identified three main mechanisms by which the macrophytic filter functions. Macrophytes either act indirectly upon particles by: (i) reducing flow velocities and initiating sedimentation (Madsen *et al.*, 2001; Schulz *et al.*, 2003; Green, 2005; Cotton *et al.* 2006; Wharton *et al.* 2006); and/or (ii) by dampening turbulence and reducing the re-suspensive shear stresses (Bulthuis *et al.* 1984; Eckman *et al.*, 1989); or (iii) macrophytes may act directly, by acting as an obstacle to particles in transportation and physically trapping sediment among their leaves and roots (Merezkho, 1973; Vermaat *et al.*, 2000; Cotton *et al.*, 2006; Wharton *et al.*, 2006). These direct and indirect processes are necessarily linked, in that changes in velocity will affect trapping potential, but their mechanisms should operate separately at a given velocity. No attempt has been made to separate the three processes, even at a single velocity, and most authors have focused on examining one aspect of the sedimentation process. Most attention has focused on vegetation effects on sedimentation while comparatively few authors have investigated effects on resuspension. Until recently, previous studies have generally referred to the likelihood of sediment trapping, but made no attempt to examine or quantify, the material directly trapped by the plants. Recent papers, arising from LOCAR, by Cotton *et al.* 2006 and Wharton *et al.* 2006 have directly examined the volume of sediment accumulated beneath macrophyte stands, and the composition of this sediment, and have helped address this knowledge gap.

In such a recent and experimental field, data collection methodologies vary widely, and are rarely comparable between non-collaborating studies. In addition, literature

from marine and lake contexts were consulted to supplement the limited number of river studies, with the result that previous research findings must be given in the context of the environment and the method used in each study. The text in this section is necessarily focused on both methods and results, while the general findings from the literature are summarised in a conceptual model in Section 6.3. Despite different methods, environments and equipment, previous empirical approaches may generally be separated into one of two broad methodologies: (i) indirect measurement: by analysing *in situ* sediments, accumulated over an uncertain time period, and relating these to the processes that have led to their deposition; or (ii) direct measurement: by investigating the sediment processes themselves.

6.2.1 Indirect measurement of sediment transport processes

Under the indirect quantification method, sediments deposited within vegetation stands are measured and compared either seasonally in the same spatial location (Welton, 1980; Schulz *et al.*, 2003), or are contrasted to vegetated areas adjacent or upstream of the vegetation (Welton, 1980; Sand-Jensen, 1998; Schulz *et al.*, 2003). Indirect methods were generally used in earlier research or as subordinate aspects of more recent research and have the principal advantages of being simple and rapid to implement and require minimal equipment and expense. They deliver simple summary measures of vegetation influence over a long time period and are assumed to give a measure of sediment deposition over a range of discharge conditions. However, the indirect observations relate to sediment accumulation over uncertain time periods; they can give no indication of sedimentation rates and are vulnerable to the influence of unknown short-term antecedent conditions. They also give gross,

‘black-box’ (Chorley and Kennedy, 1971), summary measures of sediment processes: they do not differentiate between the separate vegetation influences of increased sediment deposition, reduced resuspension or direct sediment trapping.

An early attempt to provide a quantified estimate of sediment retention by macrophytes was carried out by Dawson (1978). He describes sediment retained below *Ranunculus calcareous* stands at the end of the growing season in the River Piddle, Dorset, as being mainly organic “silt and soft sediment” of 10-20 mm mean depth (Dawson, 1978 p. 76). This estimate was given as a supporting observation in a much larger investigation focusing on the effects of macrophytes on river flows and no description was given as to the methods used to quantify the sediment depth.

Welton (1980) undertook a much more detailed reach scale study of seasonal and spatial variations of sediment retention in the Tadnoll Brook, a tributary of the River Frome. These measurements relied on visual mapping of sediments and the sampling of sediment depth by an unspecified method but with a suggested accuracy of 0.5 cm. Welton found that total quantities of both sand and organic detritus varied with discharge throughout his two and a half year study period but that, in general, at maximum plant biomass, the majority of sediment (both organic and inorganic) was associated with the margins of the stream and with beds of *Ranunculus* spp. By each November the organic detritus cover under *Ranunculus* stands had dramatically reduced (0% of total area of detritus cover November 1972, and 1% November 1973) after the senescence of the plants.

Marginal areas, in contrast, had reduced flows relative to the main channel regardless of season and retained fairly constant levels of detritus even after marginal plant cover had declined. Sand was found to be more strongly associated with *Ranunculus* stands than with the margins but showed less seasonal variation than detritus; *Ranunculus* beds still contributed 59% and 38% of the total area of sand during November 1972 and 1973 respectively. However, levels of sand cover beneath macrophytes in November were still much reduced from their May percentages (82% in May 1972, 74% in May 1973 and 77% in May 1974) (Welton, 1980). Seasonal changes in sediment retention under *Ranunculus* beds may be attributed to the seasonal growth and decline of the macrophyte stands and washout by autumn floods.

More recent, smaller scale studies into sediment retention in individual macrophyte stands were conducted by Sand-Jensen and Mebus (1996) and Sand-Jensen (1998) in lowland Danish streams. Detailed mapping of surface topography in and around vegetation stands of different vegetation species was carried out (by an unspecified method but with a stated accuracy of ± 0.5 cm) and determined that a significant increase in surface topography occurs within macrophyte beds relative to upstream areas. Mean increases within vegetation relative to the outside sediment surface height varied according to species but were found to reach as much as 11 cm for *Callitriche cophocarpa* (Sand-Jensen, 1998). Sand-Jensen also examined sediment cores from in and around macrophyte beds and found that the size composition of the upper layer of sediment was strongly influenced by macrophyte cover (Sand-Jensen, 1998). In the majority of sites, sediments in the upstream two-thirds of the macrophyte beds (an area in general agreement with that of the raised topography)

were much finer and less variable than outside the vegetation. This size distinction indicates that macrophytes may trap sediment in a size-specific manner while the range of particle sizes observed within the vegetation beds (200 – 500 μm) points to both a bed load and suspended sediment contribution to surface sediment.

Schulz *et al.* (2003) employed both direct and indirect methods and used sediment traps and sediment coring to investigate reach-scale macrophyte influences (*Sagittaria sagittifolia* (arrowhead), *Nuphar lutea* (yellow water lily) and *Potamogeton pectinatus* (fennel pondweed) on sedimentation in a lowland German stream. Sediment core stratigraphy, observed across the stream bed, indicated the presence of a surface organic layer in August that varied from 0.3 cm outside of vegetation to 8 cm within vegetation. This organic layer indicates the high contribution of the plants to the sediment they retain and it was calculated that organic matter accounted for 15 - 49% of sediment deposition during the vegetation growth period. The organic layer was observed to disappear from the study reach in October after the senescence and washout of plants.

Recent work emerging from LOCAR (Cotton *et al.*, 2006 and Wharton *et al.*, 2006) describes the pattern, volume and character of sediment trapped beneath stands of *Ranunculus calcareous* gathered from monthly measurements of macrophyte cover, sediment depth, water velocity and monthly sampling of sediments beneath macrophyte stands. Cotton *et al.* considered two sites on the River Frome: Maiden Newton in the upper reaches of the Frome (upstream of the study sites used in this thesis) and Pallington in the middle reaches of the river (downstream of the study sites used in this thesis). Wharton *et al.* considered the same sites at Maiden Newton

and at Pallington and three additional sites: East Stoke in the lower reaches of the Frome, Baggs Mill on the River Piddle and Snatford Bridge on the Bere stream (a tributary of the Piddle).

At each site a single macrophyte stand was chosen for study. Sediment depth was measured by measuring the depth of penetration of a 'fixed diameter measurement device' (of un-specified diameter). Point measurements were then extrapolated to estimate the volume of fine sediment beneath the vegetation. The sediment accumulation was reported in units of metre cubed of accumulated sediment per metre squared of plant cover ($\text{m}^3 \text{m}^{-2}$) allowing comparison of the trapping rates of different sizes of sediment stands across sites and across monthly samples. Sediment size distribution and organic matter content were sampled at six locations beneath the plant stands (one sample within the trailing end of the *Ranunculus* and five in the upstream rooted area). The sampling locations were chosen using randomly generated x and y coordinates altered every month to avoid sampling the same area. The size distributions reported by Cotton *et al.* refer to the inorganic component of the sediment only, whereas Wharton *et al.* report the 'effective sediment size' distribution for all sampled particles, both organic and inorganic, with no disaggregation of particles in pre-treatment. Cotton *et al.* measured the organic matter content of their samples through loss on ignition.

In both studies it was found that the amount of accumulated sediment varied immensely: varying within stands, between study reaches and varying throughout the summer growing season. For example, at Pallington Cotton *et al.* describe a steady increase in accumulated sediment between May and July (showing that that the

volume of sediment retained by the plants increased as the size of the plant stand increased), followed by a rapid decline in sediment accumulation from July to September as vegetation stands declined, whereas at Maiden Newton this pattern was disrupted by flood events which washed sediment from the reach (see Figure 6.1). This discharge dependency underlines the difficulties in using snapshot direct sampling of sediments to determine seasonal change.

Wharton *et al.* reported that higher volumes of sediment accumulation were recorded in the Frome catchment than in the Piddle catchment (the highest accumulations were recorded at Pallington ($0.085 \text{ m}^3 \text{ m}^{-2}$) in April 2003 and at Maiden Newton ($0.08 \text{ m}^3 \text{ m}^{-2}$) in July 2003), suggesting higher sediment availability in the Frome. Interestingly, Wharton *et al.*'s results also suggested that the downstream sites in each river accumulated less sediment than those in the middle and upper reaches (always less than $0.2 \text{ m}^3 \text{ m}^{-2}$ in any month). Wharton *et al.* suggested that inter-stand differences could be linked to overall vegetation cover in the reach and the variation in the seasonal patterns of colonisation, growth and die-back of the *Ranunculus*. Wharton *et al.* also proposed that differences in sediment accumulation within individual stands could be explained by differences in the rooted and trailing sections of the plant, with higher sediment accumulation occurring in the upstream rooted area where water velocities are lowest. Cotton *et al.* postulate that the sediment retained beneath plant stands is not static, even at non-flood flows, and suggest that sediment travel slowly through the plants stands migrating from the roots to the end of the *Ranunculus* plant. This has importance for direct measurement of sediment in that the sediment sampled at different locations through the plant may not accurately reflect the trapping rates of that location.

Cotton *et al.* found that the inorganic sediment beneath *Ranunculus calcareus* was dominated by sand (63 – 1000 μm) with silts and clays (0.37 – 63 μm) making up less than 10% of the accumulated sediment (by volume), fine sand (125 – 250 μm) was dominant in the upstream site at Maiden Newton while coarse sand (250 – 500 μm) was predominant at Pallington downstream (see Figure 6.2). Cotton *et al.* postulated that the dominance of the sand-sized fraction suggests that the majority of trapped sediment was derived from saltating fine-grained material. Wharton *et al.* measured the particle size of both organic and inorganic particles (including aggregates) and found that the dominant particle size in the trapped sediment influenced by the aggregation of particles as faecal pellets. For example, at Snatford Bridge on the River Piddle the faecal pellets were judged to range in size from 25 – 400 μm and this size range accounted for 60% of the accumulated sediment (by volume). Cotton *et al.* reported that the organic matter content of the accumulated fine sediment ranged from 9 - 106 mg g^{-2} .

These indirect research studies confirm the capacity of macrophytes to store sediment, and give some measure of the temporal changes in storage. Sediment storage was generally greatest in summer and lowest in winter but was affected on a short term basis by changes in discharge and sediment supply (Welton, 1980). It would seem that the macrophyte store was not permanent even in summer: that sediment retained by the plants at one discharge condition may have been removed and replaced under subsequent discharges (Cotton *et al.* 2006) and that even under base flow conditions the sediment moves slowly through the vegetation, from the upstream to the downstream end of the plant (Cotton *et al.* 2006 and Wharton *et al.* 2006). The sediment sampled at a specific location may not accurately reflect the

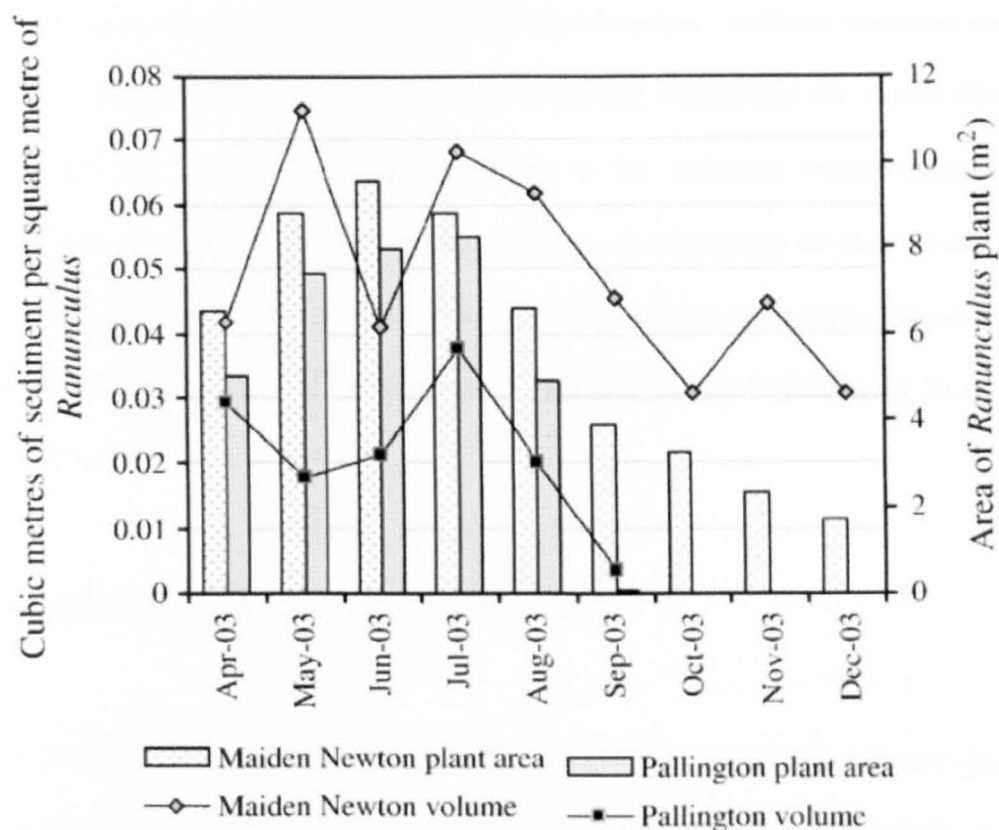


Figure 6.1 Seasonal changes in the volume of fine sediment ($\text{m}^3 \text{m}^{-2}$) of accumulated fine sediment beneath a stand of *Ranunculus calcareous* at Maiden Newton and at Pallington on the River Frome (Cotton *et al.*, 2006)

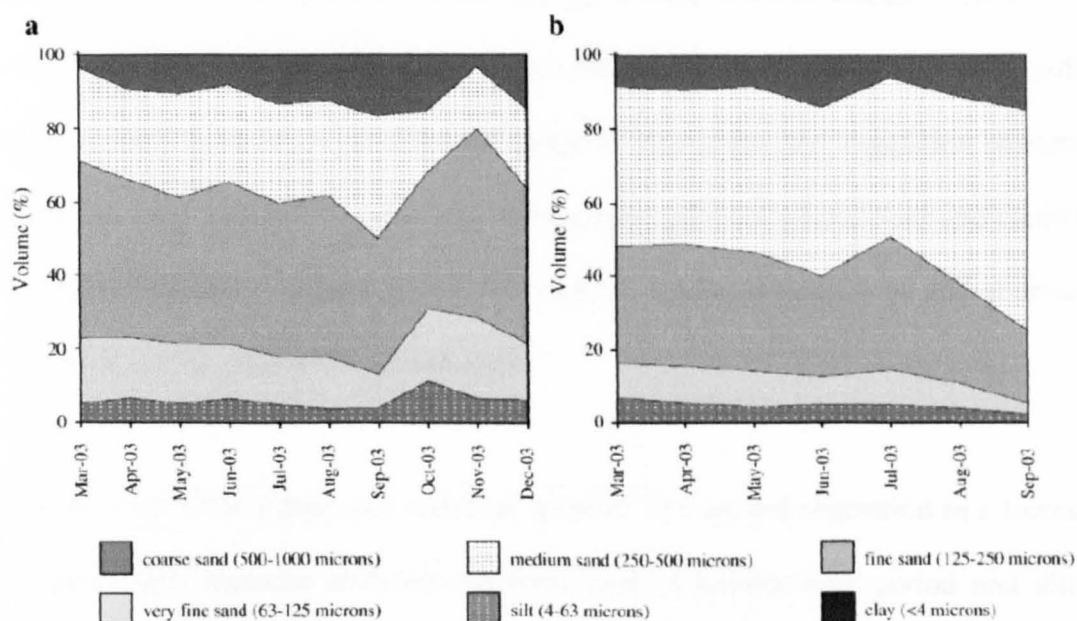


Figure 6.2 Seasonal variation in the particle size of inorganic material stored beneath a stand of *Ranunculus calcareous* at two sites on the River Frome: a) Maiden Newton and b) Pallington (Cotton *et al.*, 2006).

trapping rate or trapping characteristics at that location. Indirect measures are best used for repeat measurements over short, defined, timescales for which discharge conditions are known and may be related to the sediment measurements. The indirect methods also highlight the contribution of self-supply of plant matter to the sediment store which is not related to the plants properties as a filter. These findings have been summarised in a conceptual diagram, which is discussed in detail in Section 6.3.

6.2.2. Direct measurement of deposition rates

The methods used to measure sediment transport processes directly are generally more recent, more complex and more time consuming than that of the indirect *in situ* investigations. They have the advantage of operating over defined time periods, allowing calculation of deposition or erosion rates, but rates can only normally be calculated in terms of hours (Koetsier and McArthur, 2000) or weeks (Welton, 1980 and Schulz *et al.*, 2003). Although direct investigations often offer only a 'snapshot in time' (and generally over a limited range of discharges and vegetation changes), these can offer valuable insights into macrophyte-sediment *interactions* and could be especially valuable if repeated under a range of discharge conditions and at several times during the vegetation growth cycle.

Schulz *et al.* (2003) deployed sediment traps in and around vegetation in a lowland stream which measure sediment retention over a known time period and allow trapping rates to be determined. It was found that summer trapping rates were higher in areas downstream of and within vegetation stands relative to upstream un-

vegetated areas. For example, a large stand of the submerged plant *Sagittaria sagittifolia* was observed to produce a gradient in trapping rates along its length, with low trapping rates upstream of the vegetation ($17.5 \text{ g m}^{-2} \text{ day}^{-1}$ dry weight) and high rates downstream ($47.5 \text{ g m}^{-2} \text{ day}^{-1}$ dry weight). This would seem to indicate that higher depositional rates (and/or lower resuspension rates) occur within vegetation beds, and that this contributes greatly to sediment accumulation.

In contrast, several very detailed experimental studies of coarse particulate organic matter (CPOM) have been conducted by Koetsier and McArthur (2000). Koetsier and McArthur contrast two 100m river reaches over two seasons and employing three linked experimental designs. Firstly, they attempted a measure of 'transient CPOM' flowing into and out of individual macrophyte beds by placing nets upstream and downstream of the vegetation so that all material in transport at each location was collected. Sampling was repeated in both autumn (high littoral input) and summer (low littoral input). A second, similar, research design sought to measure transported organic matter at the reach scale in nets at the top and bottom end of the study reaches. A third method involved the release of segments of biodegradable forestry flagging tape at the upstream end of a reach to simulate fallen terrestrial leaves. These were allowed to drift for 3 hours, and then for a further 24 hours, after which the distance travelled, the type of retention barrier involved and the numbers of 'leaves' retained in the reach were recorded (Koetsier and McArthur, 2000). Afterwards, all aboveground vegetation was removed and the experiments were repeated. Differences between the two experimental runs provide a quantifiable estimate of the net effect of the macrophytes.

Results from Koetsier and McArthur's leaf analogue experiments showed that the calculated travel distances and retention coefficients of the 'leaves' were greater after plant removal in both summer and autumn, indicating that macrophytes have significant effects on retention at both high and low biomass. In the reach-scale measurement of sediment input and output, results showed that inflowing sediment volume exceeded output volume before plant clearance. After plant removal, however, differences in input and output were not significant (Koetsier and McArthur, 2000). This suggests that retention rates were greatly reduced following plant removal. Net experiments at the microscale, however, produced slightly unexpected results. For low-density beds, results conformed to theory, with inputs to the beds exceeding outputs. In high biomass beds, however, output of CPOM exceeded input. To explain this, Koetsier and McArthur describe these high biomass beds as being 'swamped' with sediment and have postulated that an upper limit of retention exists. After this limit is reached, increased hydraulic drag returns CPOM to transport (Koetsier and McArthur, 2000).

Horvarth (2004) employed similar methods to Koetsier and McArthur but his study was of much more limited scope and relied on modelled results. He conducted leaf analogue experiments over several experimental reaches, each 5 metres in length, using neutrally buoyant round paper chips. Analogues were released at the upstream end of the reaches and a screen barrier was placed at the downstream end. 'Leaves' reaching the screen after 1 hour were removed and counted and the experiment was repeated after macrophytes were removed. After each experiment the number of released and recovered particles were compared for the vegetated and vegetated states and expected particle travel distances were modelled based on a negative

exponential curve. Particle travel distances were found to be significantly shorter (approximately ten times less) with vegetation cover intact than in the same reach when macrophytes were removed (Horvarth, 2004). These results again point to macrophytes having a positive effect on sediment retention.

The direct measures of sediment deposition demonstrate the positive influence of macrophytes on depositional processes. More sediment is deposited within vegetation than outside vegetation (Schulz *et al.*, 2003) and a gradient in deposition rates can be observed with distance from the upstream boundary of submerged macrophytes. It would also appear that more deposition occurs overall in a reach that is vegetated than in the same reach when vegetation has been removed. However, the two more experimental direct measures of deposition (Koetsier and McArthur, 2000 and Horvarth, 2004) had an emphasis on large and artificial ‘organic analogues’ and these contribute little to the understanding of the effects of macrophytes on inorganic sediment particles. Inorganic particles are smaller and denser than the organic analogues used in Koetiser and Mc Arthur’s and Horvarth’s work and further research is needed to investigate any differences in their response to the macrophyte filter. The two organic studies provide some promising field methods and in this chapter the technique of releasing and measuring known sediment quantities is adapted and applied to inorganic sediment investigations.

6.2.3 Direct measurement of re-suspension rates

The majority of evidence for reduced re-suspension rates within macrophyte beds comes from literature from lentic and marine contexts (Bulthuis *et al.*, 1984; Eckman

et al., 1989; Scheffer, 1999; Vermaat, 2000; Madsen *et al.*, 2001; Kufel and Kufel, 2002). In general, macrophytic plants were found to dissipate wave energy in these environments; reducing re-suspension and improving water clarity relative to un-vegetated areas. Two main studies from a marine context provide detailed experiments and quantified evidence of reduced resuspension beneath macrophytes while others merely mention that this is likely to be the case.

Bulthuis *et al.*, 1984, used mechanical sampling methods to compare differences in suspended sediment concentrations above vegetated and vegetated mudflats. They found that the amount of suspended sediment leaving the mudflats at ebb tide was higher from the un-vegetated mudflats than from the vegetated mudflat. The ebb tide ensures that the dominant source of sediment is from resuspension of sediment from the mudflats and this implies that re-suspension was reduced by the vegetation. Eckman *et al.*, 1989, conducted a microscale study of settling rates in the marine environment. They used Plexiglas panels to collect particles settling both within vegetation and within adjacent un-vegetated areas. They found that, on average, 2.4 - 4.8 times more particulates, by weight, were accumulated within plant canopies than those in exposed areas. In a further experiment, and to separate the sedimentation and resuspension processes, they released glass tracer beads into the water column in front of the vegetation and collected the tracers on coated lab slides placed on the bed. It was found in this experiment that impaction rates of tracer particles were less under the vegetation than in vegetated areas. Relating both experiments, they argue that the greater sedimentation within vegetation beds was due to weaker shear stresses and reduced re-suspension rather than from higher deposition rates. Vermaat (2000) also describes this effect in relation to dense *Chara*

aspera beds in a lake context, theorising that the *Chara* lift the re-suspensive turbulent shear stress above the sediment surface.

The uni-directional current provided by the ebb tide in Bulthuis *et al.*'s study means it is the most comparable to the river environment, but there are still important differences between this and the freshwater lotic environment. Some evidence of reduced erosion within macrophyte beds in rivers may be inferred from results provided by French and Chambers (1996). In an investigation into habitat partitioning it was noted that trays filled with sediment and placed outside of vegetation were actively eroded, while infilling of fine sediment occurred in sediment trays within plant beds (French and Chambers, 1996). Further research in a river environment is required. In addition, studies employing direct measurement of sedimentation processes within vegetation beds have solely employed the use of gross, cumulative, measures such as sediment traps (Schulz *et al.*, 2003) and nets (Koetsier and McArthur, 2000 and Horvarth, 2004) or the calculation of particle travel distances (Koetsier and McArthur, 2000). Measurement of sediment in the water column has been restricted to net collections (McArthur and Koetsier, 2004), the measurement of light attenuation (Vermaat, 2000) and mechanical sampling (Bulthuis *et al.*, 1984). No use has been made of precision methods such as high periodicity turbidity monitoring which allows detailed measurement over very small spatial and temporal scales (Gippel, 1989; Clifford *et al.*, 1995; Clifford *et al.*, 1996). The potential for high periodicity turbidity probes to record sediment-macrophyte interactions will be explored in this chapter.

6.2.4 Biotic effects and changes to sediment biogeochemistry

Changes in the quantity, chemical composition, and grain size of sediment entering and exiting macrophyte beds may occur as a result of physical changes to the sediment and additions of organic matter while the sediment is retained within plant beds. Organic matter may be removed for utilisation by the plants themselves and by fauna associated with the plants. Plants obtain much of their nutrient requirements from the bed substratum (Barko *et al.*, 1991), while biota may remove organic matter both from the sediments (Karjalainen *et al.*, 2001) and directly from the water column (Scheffer, 1999; Kufel and Kufel, 2002). Equally, as evidenced in Section 6.2.2, the plants and biota may themselves be *sources* of organic matter. Plants in particular continually add material to the bed (Dawson, 1976) while biota may contribute organic matter and to the sediments via faecal pellets (Welton, 1980; Cotton *et al.* 2006; Wharton *et al.* 2006) and during senescence.

The nature of organic matter may also be changed through decomposition and resizing by the biota. For example, Champion and Tanner (2000) have written that nutrient processing time is much reduced in rivers of high macrophyte abundance and Cotton *et al.* 2006 and Wharton *et al.* 2006 describe the aggregation of particles in faecal pellets. Plants may even actively contribute to nutrient processing by delivering oxygen excretions to the sediments (Merezhko, 1973; Barko *et al.*, 1991; Flessa, 1994; Wigand *et al.*, 1997; Karjalainen *et al.*, 2001), which increases microbial activity, though this may be more important under oligotrophic lake conditions (Karjalainen *et al.*, 2001) than in a river environment. Biota may also play a role in the re-suspension process; bioturbation by fish and invertebrates which

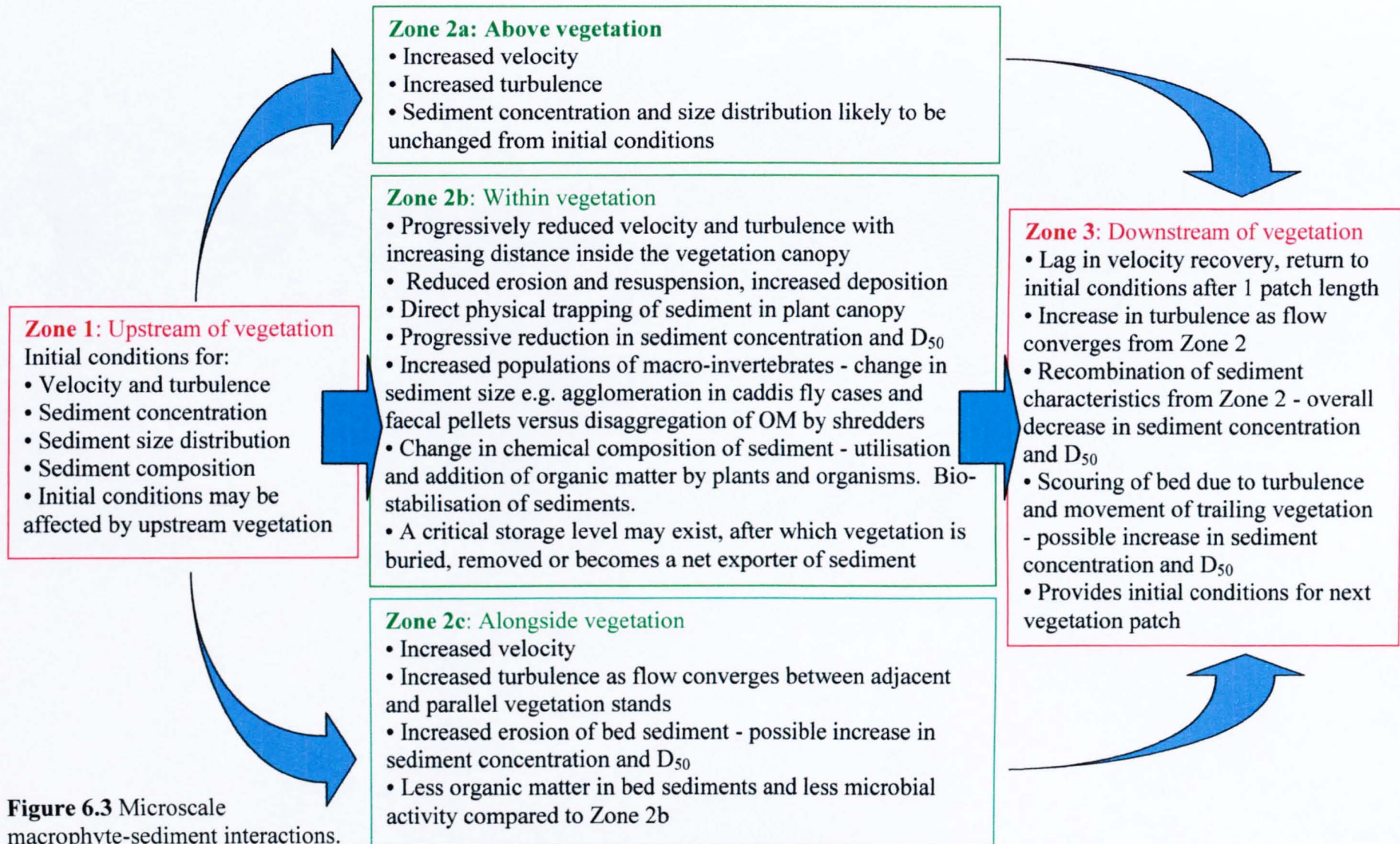
inhabit the macrophytes can affect sediments by lifting particles away from the bed and into transport (Barko *et al.*, 1991).

6.3 The conceptual model and experimental research design

The disparate nature of existing research findings and the myriad of field methodologies make it difficult to outline the general state of knowledge regarding macrophyte-sediment interactions; the conceptual diagram in Figure 6.3 was developed to consolidate the scattered information available from previous research. The conceptual diagram is based at the scale of an individual submerged macrophytic plant and is segregated into five spatial zones based on the different sediment and velocity changes likely to occur at each location. These five zones are as follows:

- Zone 1:** Upstream of the vegetation
- Zone 2:**
 - 2a:** Above the vegetation
 - 2b:** Within the vegetation
 - 2c:** Alongside the vegetation
- Zone 3:** Downstream of the vegetation

Zone 1 is positioned upstream of the vegetation and describes the initial conditions for velocity, turbulence, sediment concentration, sediment size distribution and the chemical composition of the sediment. These initial conditions are affected by changes in river discharge and external sediment supply but may also be influenced by stands of vegetation immediately upstream. Zone 2 lies downstream of Zone 1



and contains three zones located at the same downstream position but which describe the sediment and velocity changes occurring: (a) above; (b) within; and (c) alongside the vegetation. The three zones within Zone 2 recombine in Zone 3 and the sediment and velocity characteristics here are an amalgamation of the velocity and sediment characteristics inherited from Zone 2. Zone 3 then provides the downstream initial conditions for the next stand of vegetation where the processes described in the conceptual diagram will be repeated.

These five zones were used as the basis for an experimental field methodology in which high-frequency response turbidity probes were used to record the passage of introduced sediment plumes through the vegetation. Five turbidity probes were used: one upstream, one downstream and one within the vegetation and two probes alongside, to the left and right of the vegetation. Following the conceptual model, it was assumed that Zone 2a, above the vegetation, would experience unchanged sediment characteristics from the initial conditions and it was considered less important to sample this zone. The turbidity experiments were conducted within the confines of the grid-reaches at Crockways and at Hydeclift which were explored in Chapter 5. In this way, insights gained at the microscale may be linked or extrapolated to the larger grid and PT results from previous chapters. The full method and research design used for the turbidity experiments are described in Chapter 2.

Before the turbidity experiments were carried out, a preparatory ‘vegetation washing’ experiment was conducted to establish the particle size of inorganic sediment trapped by the vegetation as an essential indicator of the range of particle sizes to be used in

the turbidity experiments. As acknowledged in Section 2.10.1, the results of the washing experiments concentrate only on the inorganic component of the sediment load, as organic material was lost during pre-treatment. This method is in accordance with that of Cotton *et al.*, 2006 who analysed sediment trapped beneath macrophytes stands. Organic material has been shown by Westlake *et al.* 1972 and Wharton *et al.* 2006 to be an important component of the sediment stored beneath macrophytic plants and the results as described here are not representative of this. However, the experiments were initially conceived to support more detailed studies and are presented here as an illustration of the potential insights that could be gained and the viability of the sediment collection method.

The washing experiments cover several vegetation types not used in the turbidity experiments and revealed some interesting differences in sediment retention according to vegetation growth form. The results from these experiments are related below to justify the use of sediments in the turbidity experiments but also as findings from a preliminary experiment with its own merits.

6.4 Vegetation washing experiments

The suspected direct trapping effects of vegetation can be visually confirmed by close inspection of plant stands, but the concentration and nature of this sediment has not been quantified. In an attempt to measure the sediment load and sediment size composition of trapped inorganic particles, a simple, 'snap-shot', sampling experiment was devised. Several samples of vegetation were collected from the Crockways grid-reach in October 2003, and were examined in the laboratory with the

intention of quantifying the nature of the sediment trapped within the plant material. A full description of the laboratory procedures involved is provided in Chapter 2. The aim of this analysis was to determine the relative amounts of sediment trapped by different plant species, and to quantify the size range of the inorganic particles preferentially scavenged by the vegetation. It is acknowledged that the sediment samples represent only one point in time, and may have been heavily influenced by antecedent conditions, but it is thought that the samples provide a useful insight into the amount and size of sediment retained by macrophytic vegetation. This information was especially valuable in informing the design of the microscale turbidity experiments.

Seven vegetation samples were obtained from several plant species, each of different growth form and occupying differing positions in the channel. These included samples of emergent (*Sparganium erectum*), submergent (*Myriophyllum alterniflorum*), and overhanging riparian vegetation (*Epilobium angustifolium* (rosebay willowherb) and *Symphytum officinale* (comfrey). One sample each was obtained for *Sparganium*, *Epilobium* and *Symphytum*, while four *Myriophyllum* samples were collected to allow sampling of the full length of the plant stand. The four samples divided the *Myriophyllum* plant into roughly equal quarters and these were named M1 – M4 to denote their position from the upstream to downstream end of the patch. *Myriophyllum alterniflorum* has a longer growing season than *Ranunculus calcareous* and was chosen in preference for this experiment because of the better preservation of the plant stand. *Myriophyllum* is very similar in growth form and plant architecture to *Ranunculus* and the two may be regarded as analogous.

6.4.1 Attached sediment index

The sediment load for each vegetation sample is displayed in Figure 6.4. The estimated dry-mass sediment loads were standardised with reference to the dry-biomass of the vegetative material to obtain an ‘attached sediment index’. This was achieved by dividing the total sediment load by the total vegetation biomass and represents the average mass of sediment accumulated per gram of plant biomass and allows comparison between samples. *Symphytum officinale* has the highest attached sediment index, which, at 5.5 g of sediment per gram of dry biomass, is an order of magnitude higher than that of the other samples. *Symphytum* has a large, broad leaf with a rough surface covered in small hairs; the rough surface of the plant promotes sediment storage while the large surface area to mass ratio, afforded by the leaf structure, ensures a high attached sediment index. *Epilobium angustifolium*, by contrast, occurs in the same riparian location as *Symphytum* but has the lowest attached sediment index of the presented samples. This is presumably due to the fine smooth leaves of this plant species which are not efficient at retaining sediment. Within the channel, there is a similar contrast between the attached sediment index for the emergent *Sparganium* and the average value for the submergent *Myriophyllum*. *Sparganium* has a broad, flat, leaf structure, which is presented almost perpendicular to the flow direction. This position ensures large frictional resistance and low velocities, while the large surface area-to-mass ratio produces a high attached sediment index. *Myriophyllum*, by contrast, has very small, streamlined leaves which are an essential adaptation to its high velocity, central location. The fine, smooth, leaves of a *Myriophyllum* are less efficient at trapping

sediment and their branched shape also contributes to a low surface area to mass ratio and a low attached sediment index.

The low average value quoted for *Myriophyllum* hides a more complex picture of sediment trapping within this submergent plant. Figure 6.5 shows the attached sediment index for the four *Myriophyllum* sub-samples and illustrates a change in trapping efficiency with distance downstream. The attached sediment index gradually increases from a low at the upstream end of the plant (*Myriophyllum* 1), to a high in the third sample (*Myriophyllum* 3) and then declines again at the downstream end (*Myriophyllum* 4). There is little physiological difference in leaf size or structure between the terminal and central portions of a *Myriophyllum* plant and this suggests a dominant velocity control on sediment trapping, with a heavier sediment load found in lower velocity areas. Sand-Jensen and Mebus (1996), in a study of four submerged plant species in Denmark, found that velocity gradually decreased from upstream to downstream in a vegetation patch, reaching a minimum velocity between “one and two-thirds the distance from the upstream to the downstream end of a patch” (Sand Jensen and Mebus, 1996 p.175). After this point, velocity was observed to accelerate once again, returning to upstream levels after a distance equivalent to one patch length. These velocity observations agree well with the attached sediment load findings presented here. The increase in attached sediment index in the middle of the *Myriophyllum* plant may suggest either an increase in the total number of trapped sediment particles and/or a shift in the sediment size distribution, with an increase in the incidence of larger, heavier particles. *Sparganium erectum* also forms long stands parallel to the dominant flow direction and it may be theorised that the same sediment/velocity relation observed in

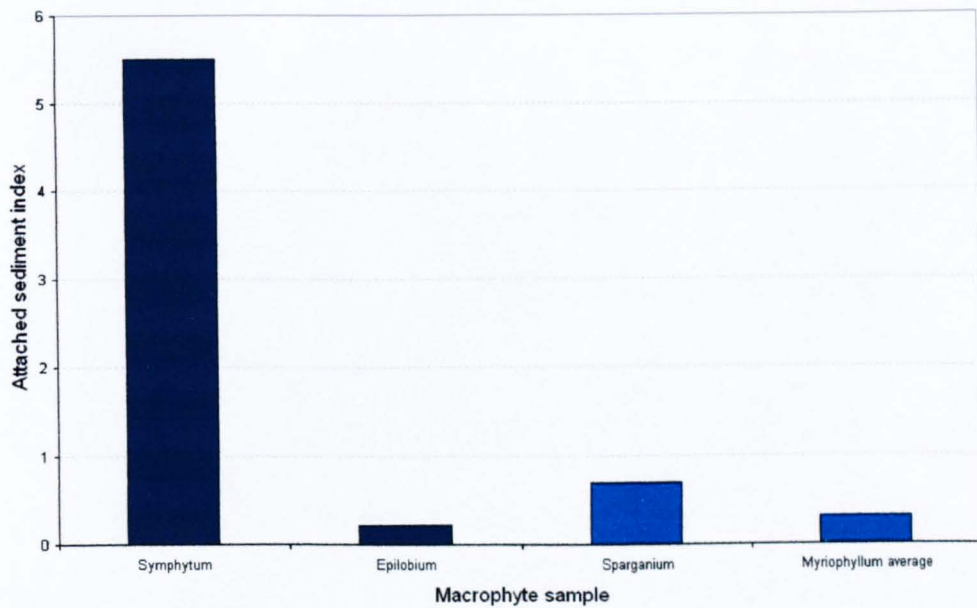


Figure 6.4 Attached sediment load for samples of four macrophyte species of contrasting growth form and channel location.

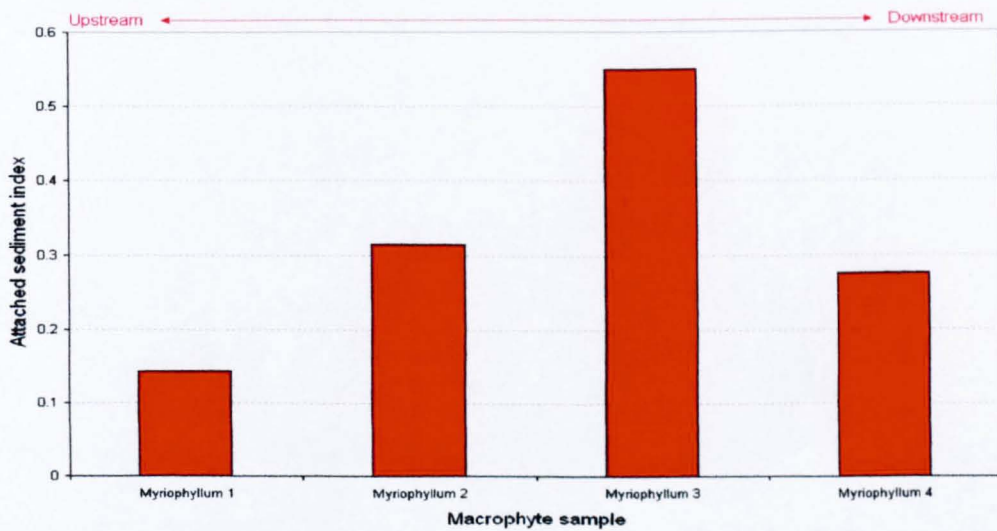


Figure 6.5 Attached sediment load for four samples of the submerged macrophyte *Myriophyllum alterniflorum*. The samples are named *Myriophyllum* 1 – 4 based on their location from the upstream to the downstream end of the plant.

Myriophyllum patches may also occur within *Sparganium* stands. The *Sparganium* sample analysed here comes from the approximate centre of a large stand.

6.4.2 Particle size analysis

Fa illustrates the mean and median particle sizes and the sediment size distribution (by volume) for the mineral fraction of the attached sediment for each macrophyte sample. The observed particle sizes may not be directly related to the attached sediment indices reported in the previous section as the attached sediment load includes both organic and inorganic material, while the particle size information considers only the mineral fraction. For the purpose of this analysis, organic particles are assumed to follow the same trends in size distribution as the mineral fraction.

To aid interpretation, the samples were initially split into two groups: (i) riparian; and (ii) in-channel vegetation. Figure 6.6b displays the particle size results for the two riparian species and demonstrates a broad agreement in the overall size range of trapped particle for both species. Figure 6.6a indicates that they have a similar mean and median particle size. However, the pattern of the distribution differs slightly. *Symphytum* displays a uniform size distribution, indicating poorly sorted sediment deposits and this reflects both the high storage potential of the rough leaf surface and also the manner of deposition. The sediment found in the riparian vegetation is likely to have been deposited by the small flood event occurring a week previously and represents wholesale deposition of sediment as water levels fell in the falling limb of the hydrograph (see Figure 2.11, Chapter 2). *Epilobium* has a more variable

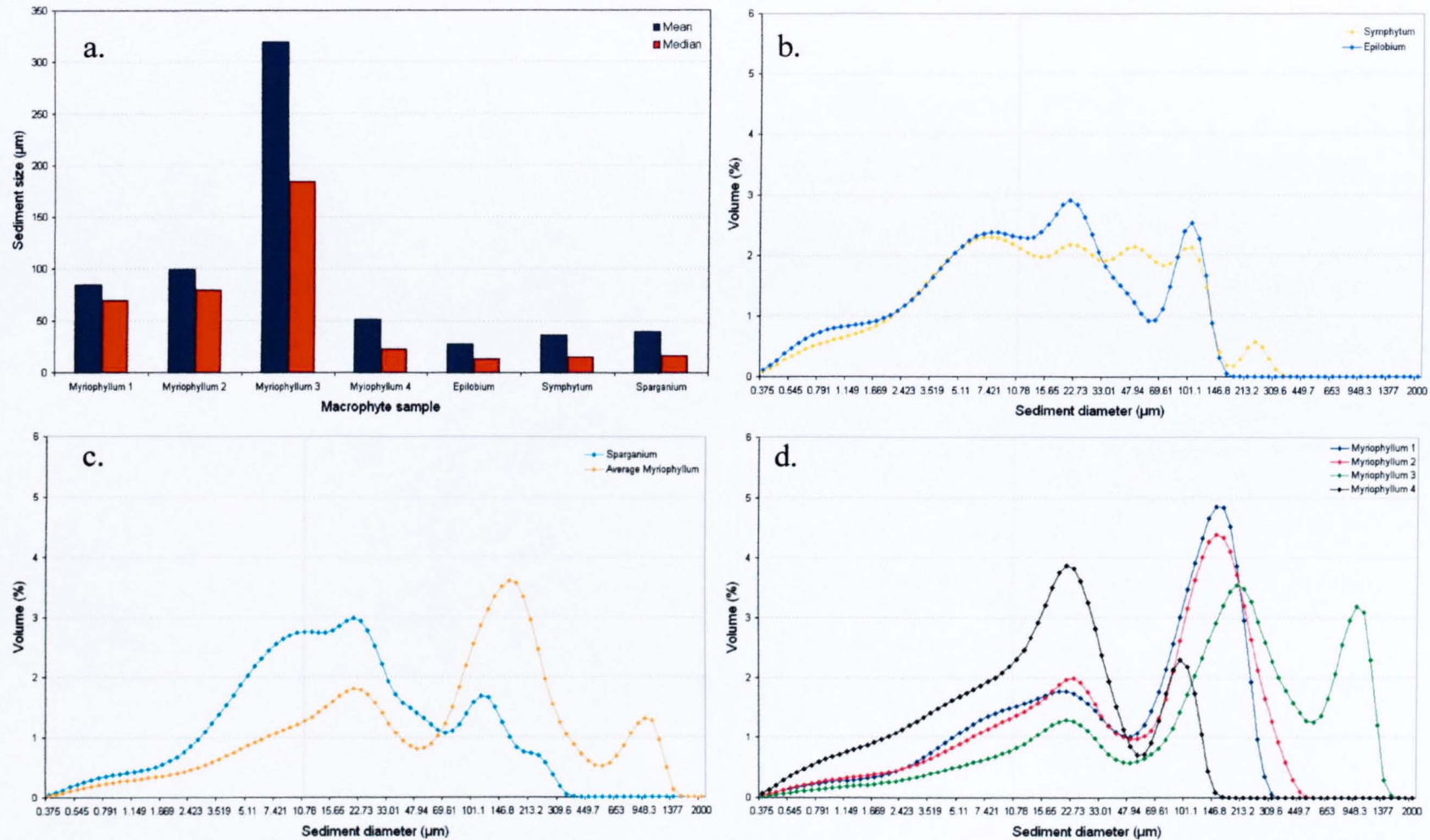


Figure 6.6 Size distributions for vegetation samples: **a)** the mean and median particle sizes and the sediment size distribution (by volume) for the mineral fraction of the attached sediment for each macrophyte sample; **b)** particle size distributions by volume for the two riparian species; **c)** sediment size distributions by volume for the in-channel vegetation species; and **d)** individual size distributions for the *Myriophyllum* sub-samples.

sediment size distribution, despite the similar manner of sediment deposition, and this may reflect the smaller, smoother leaves of this plant.

The sediment size distributions for the in-channel vegetation species, *Sparganium* and the average for *Myriophyllum*, are displayed in Figure 6.6c and demonstrate a marked difference in the size composition of sediment trapped by these two species. The *Sparganium* sample is dominated by smaller particles (modal value 21.7 μm), while the *Myriophyllum* average sample is dominated by larger particles (modal value 140.1 – 203.5 μm) and covers a much higher range of particle sizes. This size difference reflects the position of the plants in the channel and the associated flow conditions. *Sparganium* occupies marginal areas of the channel and receives only very fine sediment particles in transport due to lower marginal water velocities, while *Myriophyllum* occupies a central position, experiences higher velocities, and receives coarse sediment particles transported as suspended sediment but also with an influence from saltation and bedload. The double peak in the *Sparganium* distribution may possibly reflect differences in sediment load between the lower, submerged portion of the plant and the higher plant parts which are not often exposed to flow. The more variable sediment distribution for the *Myriophyllum* average reflects the four different distributions found in the *Myriophyllum* sub-samples.

Figure 6.6d shows the great difference in mean and median grain size between the four *Myriophyllum* sub-samples and this is borne out in the individual size distributions in Figure 6.6a. *Myriophyllum* 1 and 2, the two most upstream samples, share a very similar size distribution range and pattern; they both possess a similar

two-peak distribution, peaking at (i) 21 and 23 μm and (ii) 147 μm , and a similar mean and median grain size. This similar size distribution means that the increase in attached sediment load observed in Section 6.4.1 is most likely due to a general increase in particle retention and not caused by a shift in the sediment size distribution. The third *Myriophyllum* sample retains two peaks at similar diameters observed in *Myriophyllum* 1 and 2, but also has an additional larger peak at 948 μm . This third peak has a large influence on both the mean and median grain-size for the sample (Figure 6.6a) and may also influence the attached sediment load. Section 6.4.1 reports that the attached sediment load is highest in *Myriophyllum* 3 and the evidence from the particle size analyses suggests that this increase is due to both a general increase in the number of particles retained and a shift in the sediment size distribution towards larger, heavier particles. *Myriophyllum* 4 yielded the smallest average grain size of the four sub-samples (Figure 6.6a), in Figure 6.6d two major peaks are again evident in the distribution trace, and at broadly similar particle diameters, but the relative dominance of each peak has changed. Whereas in the three upstream samples, the first peak ($\sim 20 \mu\text{m}$) was dominant, the fourth sample sees a reversal of this with the second peak ($\sim 160 \mu\text{m}$) as dominant. This reversal explains the low mean and median grain size evident in Figure 6.6a. The third peak at 948 μm , observed in the *Myriophyllum* 3 sample, does not appear to be present in *Myriophyllum* 4. The analysis in Section 6.4.1 suggested that the attached sediment load was greater in *Myriophyllum* 4 than in *Myriophyllum* 1, and given the reduction in particle size, it must be surmised that the number of particles retained was greatly increased.

The presence of the largest grain sizes in sub-sample 3 is slightly surprising. Given the higher critical velocities required to maintain them in transport, large sediment particles should be the first to fall from suspension, and it was expected that large grain sizes would be more strongly represented in the first and second sub-samples where velocity first starts to decline. However, the washing experiments measure the particles trapped within the plant *canopy* and are not a comprehensive measure of the grain sizes removed from suspension. Lower velocities in *Myriophyllum* 3 may allow larger particles to be retained by the plant canopy, while in faster velocities, upstream and downstream, the greater turbulence and movement of plant stems, causes the particles to fall to the river bed. It is unlikely that the downstream trend in size distribution demonstrated by the washing experiments will be repeated in the bed sediments beneath the plant.

The *Myriophyllum* sub-samples indicate three ranges of sediment particle sizes, which are preferentially scavenged by the vegetation: two of these size-ranges are present, to a greater and lesser extent, in all four sub-samples while one is only evident in the sub-sample corresponding to expected minimum velocity. These three overlapping size ranges may be best delineated by quoting the three approximate modal values which mark the peaks of their distribution: (i) $\sim 20\ \mu\text{m}$; (ii) $\sim 160\ \mu\text{m}$ and (iii) $\sim 950\ \mu\text{m}$. These three distributions could be a replication of the particle size distribution of source material in the river; a response to differential particle trapping by different parts of the plant e.g. the leaf versus the stem; or perhaps, may represent a vertical zonation in trapped sediment size within the plant from the water surface to the bed, which may itself reflect the height of suspension of different particles in the ambient flow stream and the frequency of suspension.

Overall, the washing samples indicate that sediment capture by macrophytes is highly complex and that variation arises from many linked variables, including: the position of vegetation in the river; discharge conditions and the sequence of antecedent events; the height of the vegetation in the water column; the dominant suspension mechanisms in a reach and the frequency of suspension and macrophyte species differences. The next section seeks to examine these linked factors further by directly measuring sediment processes. To help inform the process-based turbidity experiments, the observations for the *Myriophyllum alterniflorum* samples in Section 6.4 have been summarised in a conceptual diagram in Figure 6.7. The diagram describes changes in velocity, sediment load and particle size along the length of the vegetation stand, with reference to the initial conditions found in the first, most upstream, *Myriophyllum* sub-sample.

The washing experiments also provided information as to the range of sediment sizes to be used in the turbidity experiments. Four sediment grades were chosen: 0 - 1 phi (1.0 – 0.5 μm), 1 - 2 phi (0.5 – 0.25 μm), 2 - 3 phi (0.25 – 0.125 μm) and ~ 3 - 4 phi (0.125 – 0.0625 μm). These correspond to ‘coarse’, ‘medium’, ‘fine’ and ‘very fine’ sand particles on the Wentworth scale. Collectively, these four sediment types cover the dominant sediment size range obtained for submerged macrophytes in the vegetation washing analyses.

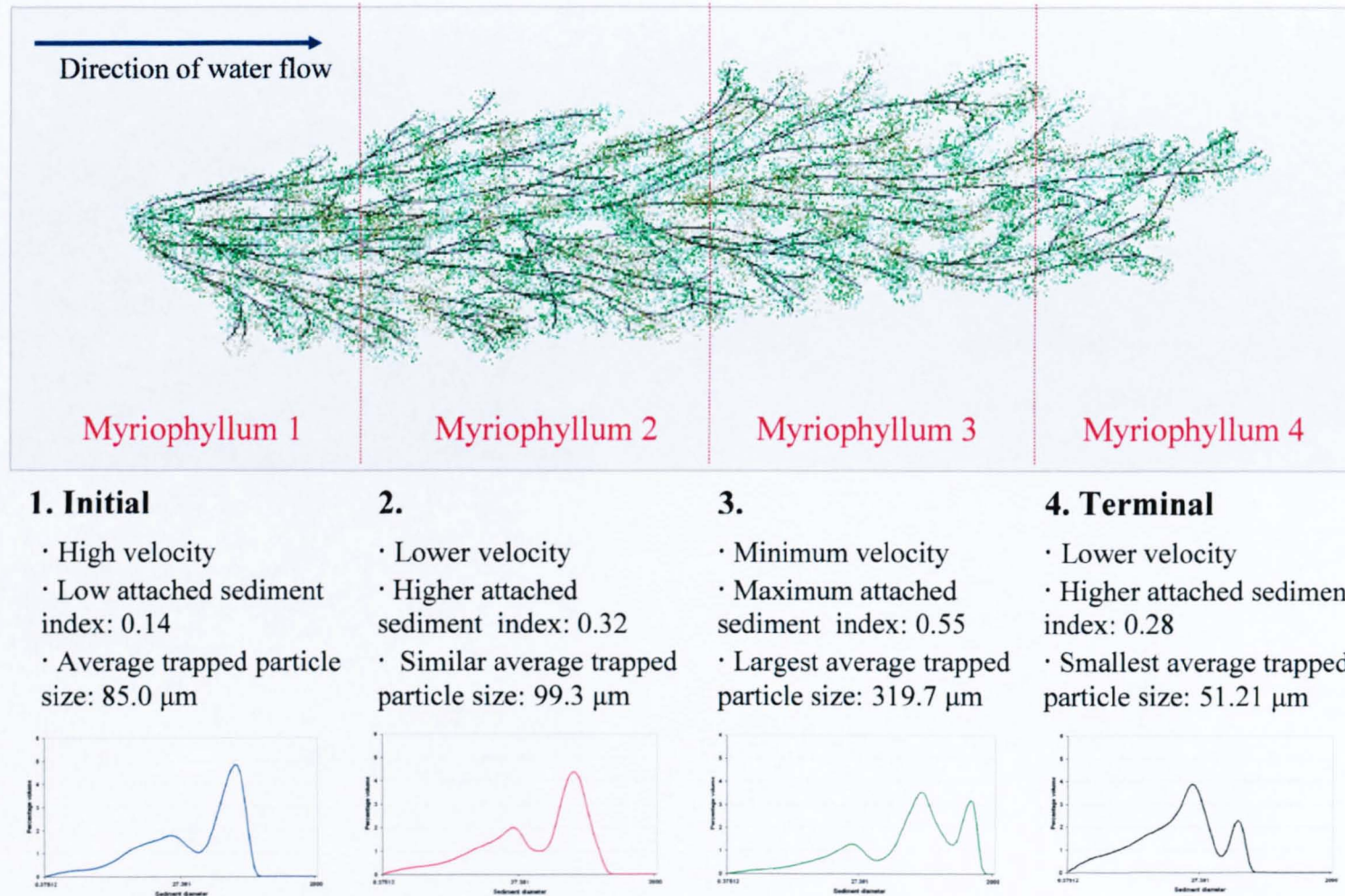


Figure 6.7 Observations for the *Myriophyllum alterniflorum* samples summarised in a conceptual diagram which describes the change in sediment load and population with increasing distance downstream.

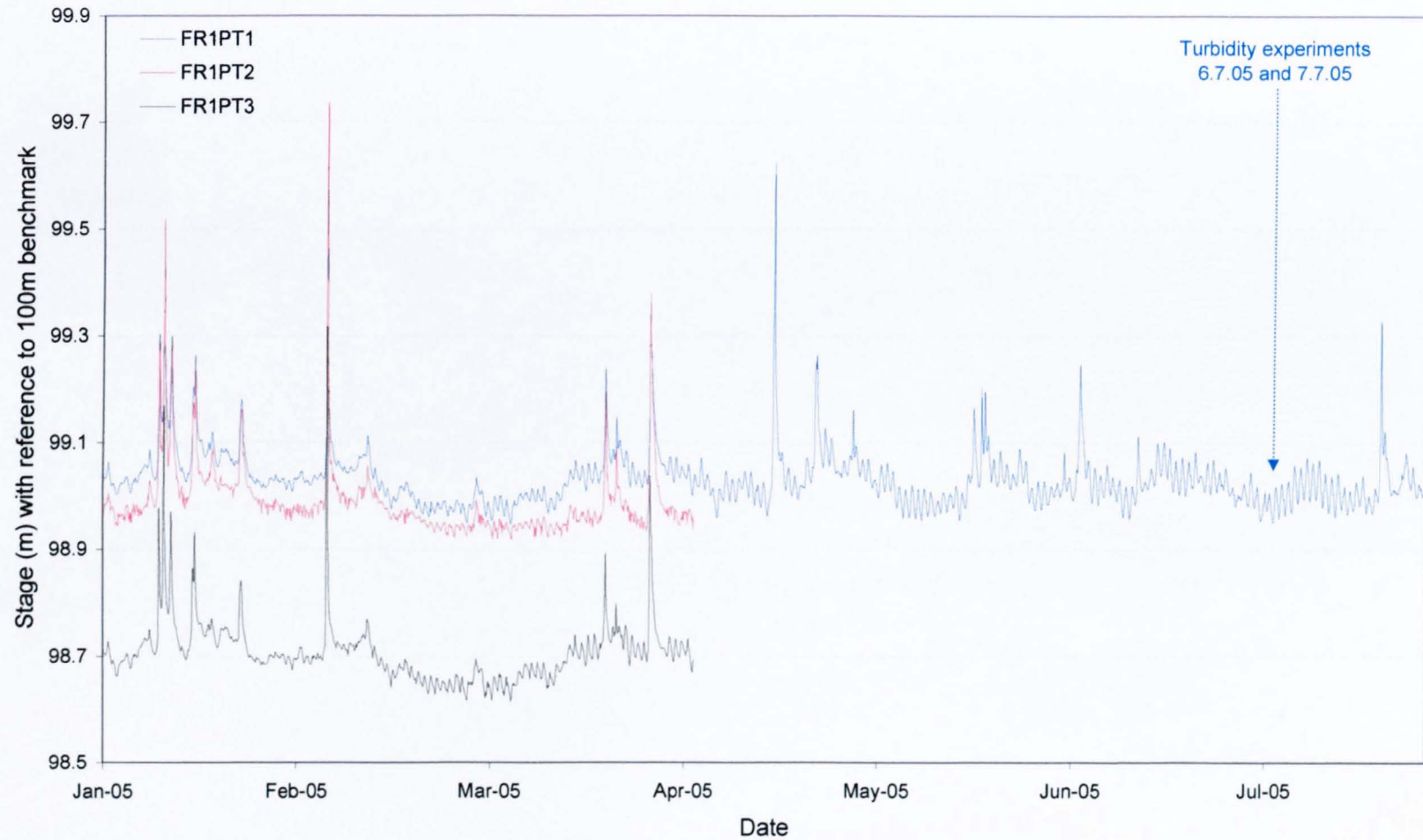


Figure 6.8 Dates of turbidity experiments at the Crockways and Hydeclift grid-reaches with reference to stage conditions at Crockways.

6.5 Turbidity experiments: the ‘Sediment hydrographs’

The next section examines a series of experiments conducted in July 2005 (see Figure 6.8) which aimed to capture the influence of vegetation on sediment in transport. In these experiments, five turbidity probes were placed upstream, downstream, within and alongside vegetation stands, to record the influence of vegetation on the passage of an introduced sediment plume. It was hoped that the turbidity probes would discern modifications made by the vegetation to either the sediment concentration or the speed of travel of the sediment pulses. The passage of suspended sediment pulses through a number of vegetated patches were compared to each other and to an un-vegetated control experiment to help account for any dispersion effects. The field method for the turbidity experiments has been comprehensively described in Section 2.10.2 of Chapter 2.

The individual turbidity meter responses to the introduced sediment events were treated as ‘sediment hydrographs’. The variable under consideration differs from that of a regular storm hydrograph, but the basic idea remains the same: measurement of a variable at a specific point and over the duration of a specific event. Similar parameters were estimated for the sediment hydrographs as are generally delineated for a regular storm hydrograph. Thus, five parameters were initially considered, either individually, or in combination, namely: (i) the total sediment throughput of an event in mg/l or ‘sum concentration’; (ii) the duration of the event or ‘base width’ of the hydrograph; (iii) the highest sediment concentration measured in each event or ‘peak concentration’; (iv) the time taken to reach peak concentration or ‘time to peak’ (T_p); and (v) the ‘time to recession’ (T_r). Table 6.1

provides a precise definition of how each parameter was obtained, while Figure 6.9 provides a visual illustration of the parameter delineation in practice.

The underlying, low-level, variation present in the turbidity series meant that a more objective method of delineating peak duration was required. This was achieved by applying a simple ‘split-window’ averaging procedure to each turbidity series to identify where discontinuities and abrupt changes were present. The sampling ‘window’ refers to the number of observations considered in a single iteration of the formula, and this window may range in size depending on the series characteristics and the problem under consideration. The procedure used in this analysis was based on the following formula from Harmar (2004):

$$A = (X_1 - X_2)^2 \tag{6.1}$$

where:

A = Variation in the mean of the series

X_1 = Mean sediment concentration in window 1

X_2 = Mean sediment concentration in window 2

This formula compares the mean sediment concentration characteristics from the first window to that found in the second window and the output from the procedure can be evaluated visually in a line graph, with the largest values of A occurring where discontinuities are greatest. Figure 6.30 shows an example plot of a measured trace of sediment concentration and the corresponding values of A , based on a window size of 100 observations, and shows the good agreement between the location of high

Table 6.1 Description of the parameters used to quantify the turbidity response or 'sediment hydrographs' resulting from introduced sediment events.

Parameter name and measurement units	Definition	Describes
Sum concentration (mg/l)	The total sediment throughput of an event; calculated as the sum of all turbidity readings minus total background turbidity levels.	Peak magnitude
Base width (seconds)	The time duration of the sediment event, or 'width' of the sediment hydrograph; defined as the time interval between corresponding peaks in the split window averaging procedure.	Peak shape
Peak height (mg/l)	The highest sediment concentration measured in each event, minus the average background turbidity.	Peak magnitude
Time to peak (Tp) (seconds)	Time interval between event initiation to peak concentration – defined as the time interval between the first peak in the split window averaging procedure to the time of peak concentration.	Peak shape
Time to recession (Tr) (seconds)	Time between the event peak concentration and the event termination – defined as the time interval between the point of peak concentration to the last peak in the split window averaging procedure.	Peak shape
Tp/Tr (ratio value)	A measure of peak symmetry; obtained by dividing time to peak (Tp) by time to recession (Tr).	Peak shape

values of A and the abrupt changes in sediment concentration which mark the beginning and end point of each peak. The size of the sampling window may be altered by the user to best describe a specific data set and this introduces some subjectivity into the analysis. To account for this, a sensitivity analysis was carried out on three turbidity series of different pulse magnitude and shape. Five appropriate window sizes were chosen based on initial estimates of the general time spacing between pulses and the base width of the sediment pulses themselves. These were based on five multiples of the 0.2 second time unit which separates each individual observation: 50; 75; 100; 125; and 150. A window size of 100 observations (or a time-length of 20 seconds) was chosen as the optimum window size and provided sufficient detail to accurately delineate the major discontinuities in the series but was also of sufficient breadth to smooth out smaller discontinuities occurring within and between the sediment pulses. As all the turbidity experiments were set up in the same way, with a defined time interval between sediment pulses and a set sediment volume in each pulse, it was decided that one common window size could be used for each turbidity series. This provided a standardised and objective assessment of all series.

6.5.2 Presence and absence of a coherent response

The first stage of the turbidity analysis focused on the identification of peak 'presence' and 'absence' in each experiment using the split-window averaging procedure. All but one experiment showed some turbidity response in the most

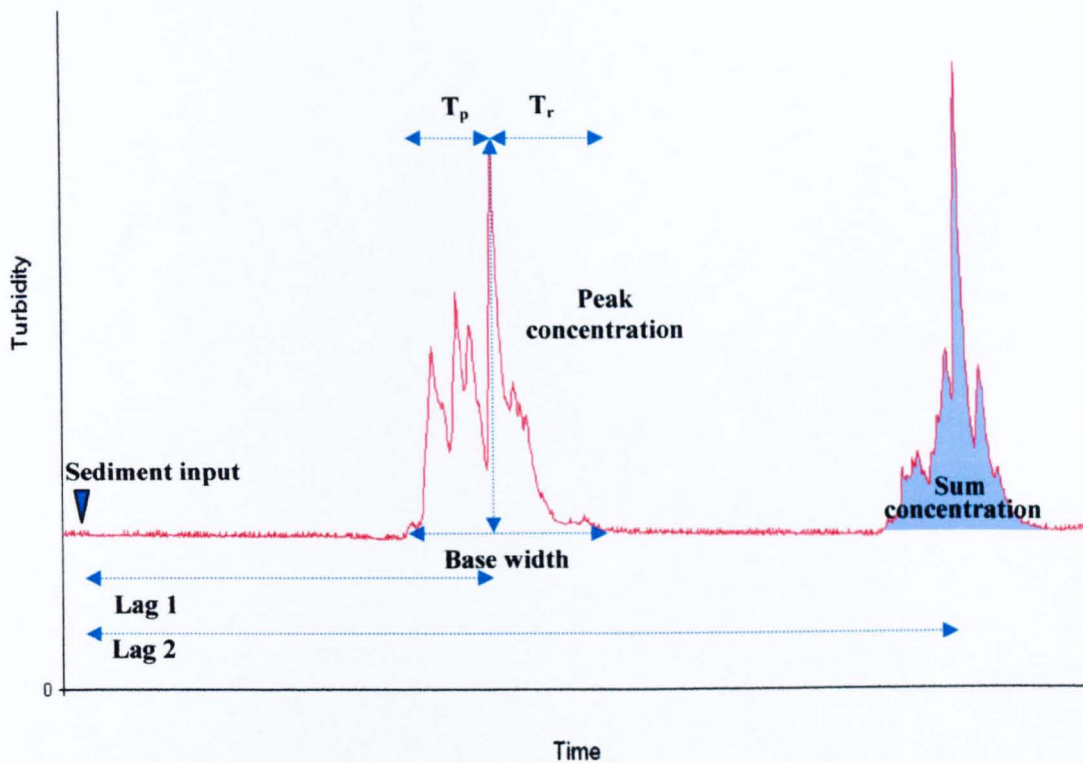


Figure 6.9 Visual illustrations of the ‘sediment hydrograph’ parameter delineations in practice: ‘sum concentration’; ‘base width’; ‘peak concentration’; ‘time to peak’ (T_p); and ‘time to recession’ (T_r).

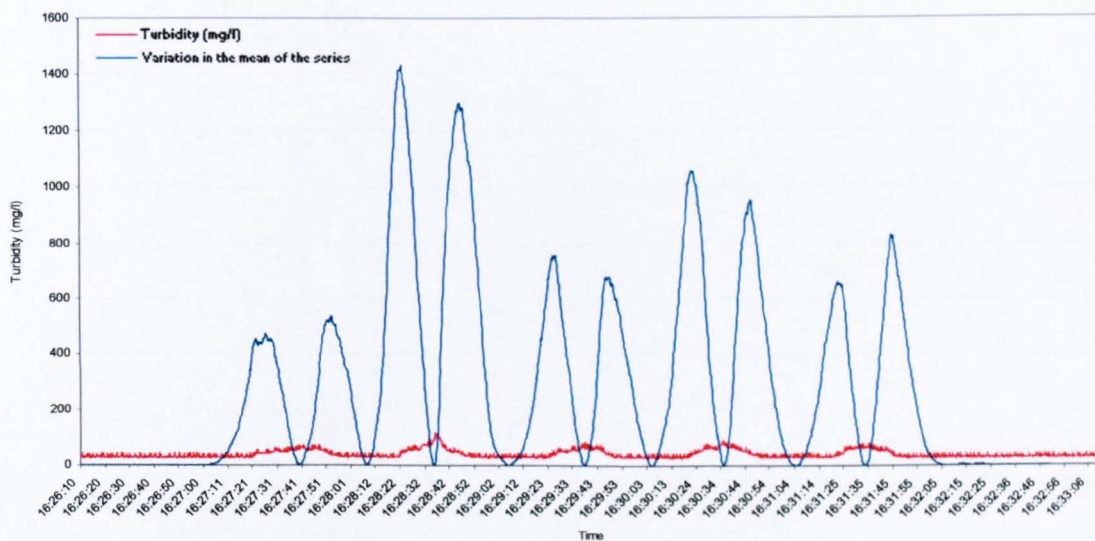


Figure 6.10 An example plot of a measured trace of sediment concentration (T6) and the corresponding values of A , based on a window size of 100 observations. This shows the good agreement between the location of high values of A and the abrupt changes in sediment concentration which mark the beginning and end point of each turbidity peak.

upstream turbidity probe (T1), but responses at more than one probe were necessary to provide a means of comparison between probe locations. Therefore, experiments that did not display a coherent response in any of the four downstream probes (T2 – T5) were not considered for further analysis. The results of this initial presence and absence testing are shown in Table 6.2 and this demonstrates that 11 out of the 25 experiments showed some downstream response. This testing was not just a means of data reduction, but also provided information in itself. The ‘discarded’ experiments are not without value; they accurately describe the effects on sediment concentration during the experiment, in that the sediment signal is removed over a very short time and space. The lack of response in probes downstream of T1 does not indicate that vegetation has no effect on sediment transport. On the contrary, it may suggest that the vegetation effect is ‘total’ and that none of the sediment is observed beyond the upstream vegetation margin. The almost universal positive response to sediment events at T1 suggests that velocity reduction upstream of the plants may cause particles to fall from suspension after passing through the upstream probe but before they reach any of the downstream probes. For example, Plate 6.1 shows the upstream portion of the medium vegetation patch used in the turbidity experiments at Hydeclift; this experiment was carried out under the lowest velocity conditions (Table 6.2) and clearly demonstrates the rapid deposition of the coarser sediment grades before the vegetation margin. The 0-1 phi (1000 – 500 μm), sediment appears to fall instantaneously from suspension, possibly without influence from the vegetation, while the 1-2 phi (500 – 250 μm) and 2-3 phi (250 – 125 μm) sediment have a longer travel distance but appear to be greatly affected by the presence of vegetation.

Table 6.2 Description of the 25 micro-scale turbidity experiments from July 2005. The use of different vegetation sizes was intended as a seasonal analogue while the different sites enabled investigation into the effects of changing depth. Velocity was partly conditioned by upstream vegetation.

Experiment	Site	Vegetation	Sediment	Water depth	Velocity	Locations at which peaks evident	Analysis
1	Crockways	None	3 – 4 phi	46	61.3	T1, T2, T3, T4 and T5	Yes
2	Crockways	Small	0 – 1 phi	45	79.5	T1 only	Yes
3	Crockways	Small	1 – 2 phi	45	79.3	T1 only	
4	Crockways	Small	2 – 3 phi	45	80.7	T1, T2, T4 and T5	
5	Crockways	Small	3 – 4 phi	45	81.5	T1, T2, T3, T4 and T5	
6	Crockways	Medium	0 – 1 phi	53	66.5	T1 only	Yes
7	Crockways	Medium	1 – 2 phi	53	66.0	T1 only	
8	Crockways	Medium	2 – 3 phi	53	60.9	T1 only	
9	Crockways	Medium	3 – 4 phi	53	76.7	T1 and T2	
10	Crockways	Large	0 – 1 phi	49	53.9	T1 only	Yes
11	Crockways	Large	1 – 2 phi	49	56.4	T1 and T2	
12	Crockways	Large	2 – 3 phi	49	69.0	T1, T2 and T3	
13	Crockways	Large	3 – 4 phi	49	53.1	T1, T2, T3 and T4	
14	Hydeclift	Small	0 – 1 phi	20	89.6	No peaks evident in any probe trace	Yes
15	Hydeclift	Small	1 – 2 phi	20	92.5	T1 only	
16	Hydeclift	Small	2 – 3 phi	20	95.4	T1 only	
17	Hydeclift	Small	3 – 4 phi	20	90.4	T1, T2 and T4	
18	Hydeclift	Medium	0 – 1 phi	17	22.7	T1 only	Yes
19	Hydeclift	Medium	1 – 2 phi	17	20.4	T1 only	
20	Hydeclift	Medium	2 – 3 phi	17	20.7	T1 only, perturbations in T2, T4 and T5	
21	Hydeclift	Medium	3 – 4 phi	17	24.0	T1, T2, T3, T4 and T5	
22	Hydeclift	Large	0 – 1 phi	19	42.3	T1 only	Yes
23	Hydeclift	Large	1 – 2 phi	19	45.4	T1 only	
24	Hydeclift	Large	2 – 3 phi	19	44.1	T1, T2, T3 and T4	
25	Hydeclift	Large	3 – 4 phi	19	43.8	T1, T2, T3 and T4	

Table 6.2 also reveals that the response of downstream turbidity probes in each experiment is heavily conditioned by sediment type. In general, the two finer sediment grades, 3 - 4 phi and 2 - 3 phi, showed the greatest level of response; peaks were detected in downstream probes for all six of the 3 – 4 experiments and for half of the six 2 – 3 phi experiments. The two coarser sediment grades provided a much more limited response downstream; the 1-2 phi sediment showed a downstream response in only 1 experiment, while the coarsest sediment, 0-1 phi, showed no downstream response in any of the six experiments. As outlined earlier, the selection of the four sediment grades was based on the size distributions evident from the washing experiments in Section 6.4. The coarser sediment grades were present in the vegetation washings, even in sub-sample 4 of the *Myriophyllum* plant, while similar sediment sizes produced little response downstream of T1 in the turbidity experiments. These different observations suggest one of three things: (i) that there are significant differences in the trapping efficiency of *Myriophyllum* and *Ranunculus* plants; (ii) that coarser sediments enter the plant canopy predominantly as bed load; or (iii) that coarser sediments enter the plant canopy as suspended sediment but under higher discharges than that experienced during the July turbidity experiments.

6.6 The clear water control: implications for the vegetated experiments

Table 6.2 identifies the 11 experiments that demonstrate a coherent turbidity response downstream of T1. The first of these experiments is a ‘clear-water’ control

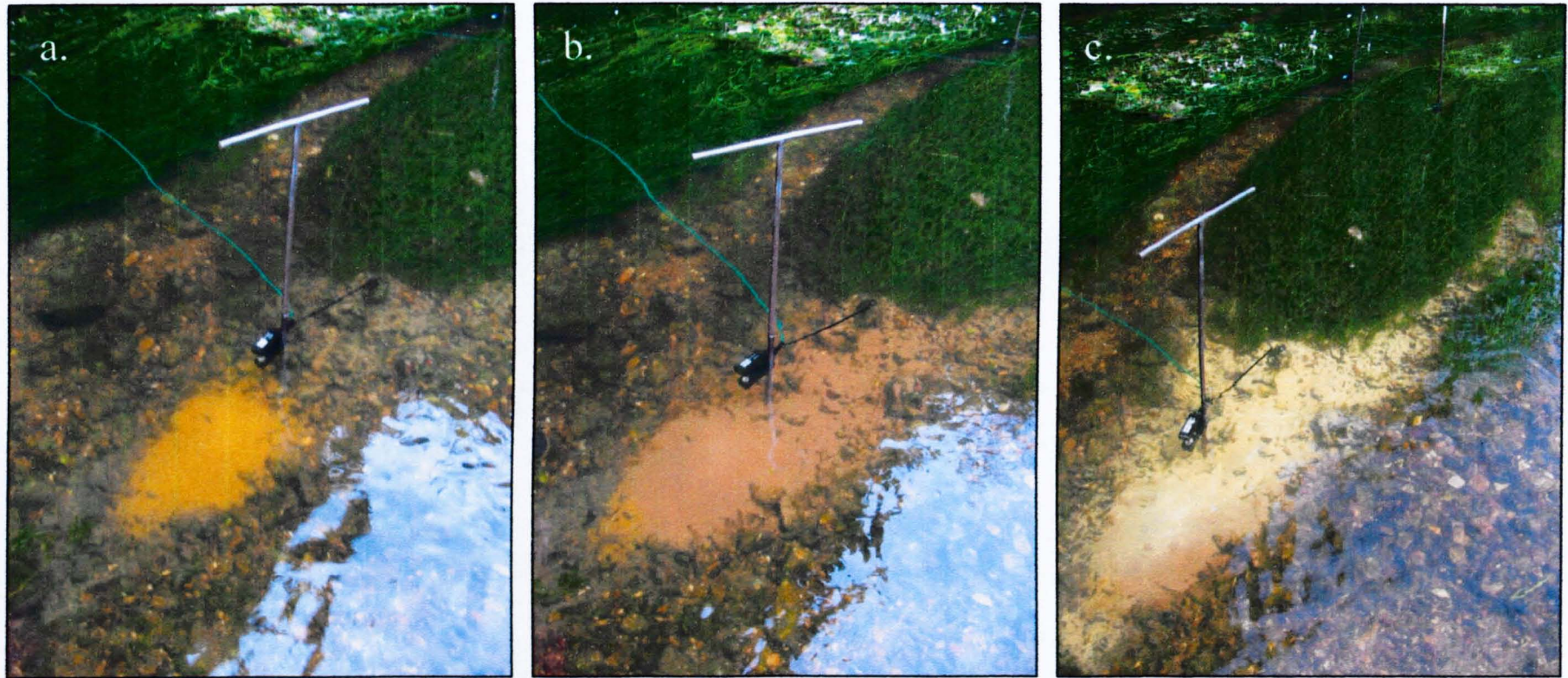


Plate 6.1 Photographs illustrating the upstream portion of the medium vegetation patch used in the turbidity experiments at Hydeclift: **a)** 0- 1 phi sediment; **b)** 1 – 2 phi sediment and **c)** 2 – 3 phi sediment. This experiment was carried out under the lowest velocity conditions and the photographs clearly demonstrate the rapid deposition of the coarser sediment grades before the vegetation margin.

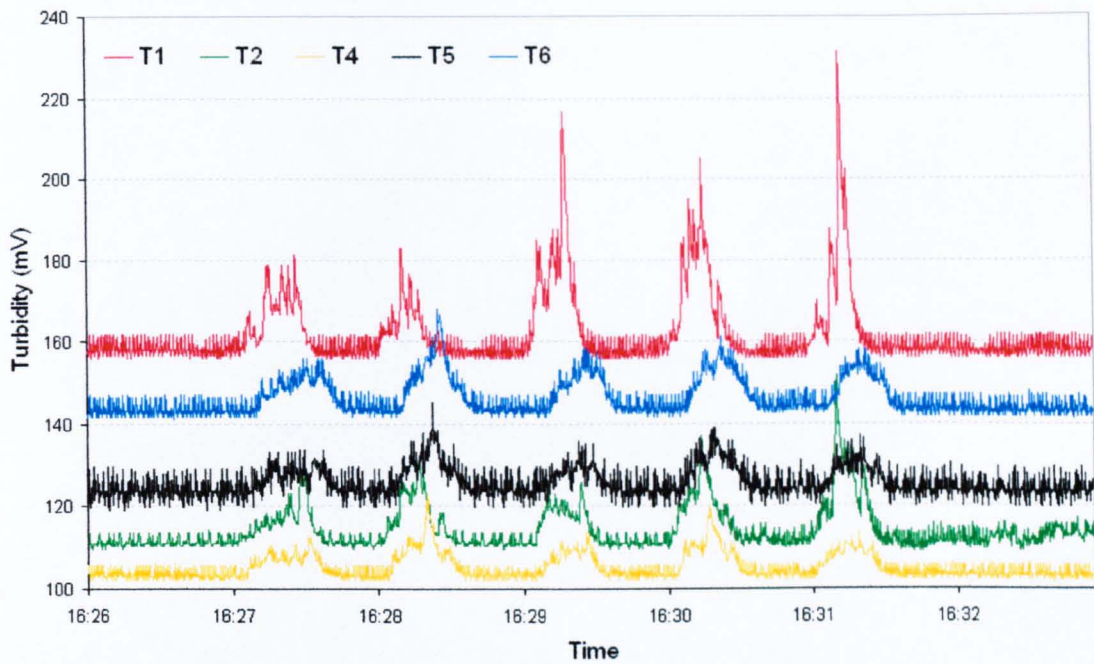
experiment designed to provide a reference for the vegetation experiments. This experiment demonstrates the changes in turbidity signals over a sample-length similar to that used in other experiments, but without the influence of vegetation. Any differences observed between turbidity traces in the control experiments are the result of dispersion processes and these must first be characterised in the absence of vegetation, in order to isolate the vegetation effects in later experiments.

The clear water experiment was carried out using 3 - 4 phi sediment in an un-vegetated area of the Crockways grid-reach, and involved the use of five turbidity probes (Ta – Te). The probes were placed in a line parallel to the channel bank line, and at one-metre intervals over a four-metre total travel length, in order to record the downstream progress of five introduced sediment events. The resulting turbidity traces were calibrated to a common background turbidity level (for calibration method see Chapter 2) and are illustrated in Figure 6.11. This background calibration allowed a comparison of sediment pulse characteristics between different locations and different instruments and was carried out for all turbidity experiments.

6.6.1 The clear water control: changes in event magnitude

Figure 6.12 illustrates the changing magnitude of the introduced sediment concentration for five individual simulated events as recorded by the turbidity probes in the clear water control experiment. Figure 6.12a shows magnitude expressed by peak concentration and Figure 6.12b expresses magnitude as the sum concentration of the event. The average sediment concentration for the five events has also been

a.



b.

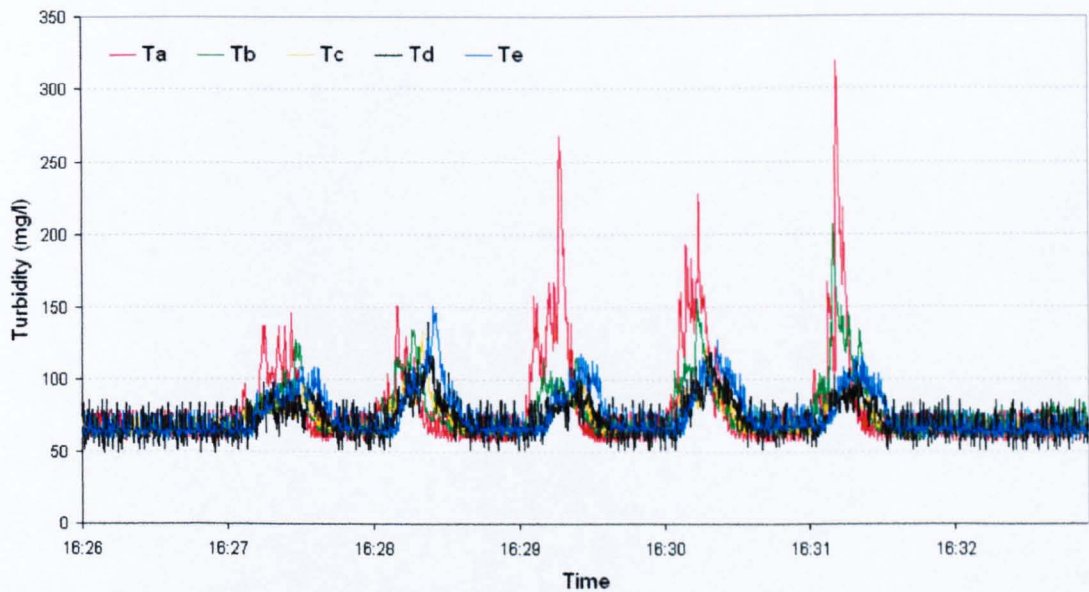


Figure 6.11 Turbidity traces for the clear water control experiment at Crockways, recorded as: a) raw millivolt data; and b) calibrated data in mg/l with a common background level (calibrated using laboratory derived calibration curves – see Section 2.10.3, Chapter 2).

shown as a reference. Given the time and distance changes between successive probes, the introduced sediment was expected to become more dispersed, and/or increasingly lost from suspension, with increasing distance downstream. It was also anticipated that a simple, declining, trend would be observed in the sediment concentration recorded by successive probes, but this does not appear to be the case, and, for some probes, the response to each event varies widely. Summary statistics describing the event response of each probe (based on sum concentration) are given in Table 6.3. On this basis, the probe responses may be divided into two groupings which display a 'consistent' and an 'inconsistent' response to the five events. These groupings are differentiated using three indicators: (i) the variation in event magnitudes at each individual probe; (ii) the comparative response to events between probes; and (iii) changes in the relative magnitude of events between probes.

Probes Ta and Tb record an inconsistent response to the five successive events. At each probe, the recorded magnitude for the individual events varies markedly. Most noticeably, both probes have a very large standard deviation and coefficient of variation in event response: 2148.29 (Cv 42 %) at Ta and 1862.51 at Tb (Cv 54 %), despite the standardised volume of sediment released in each event. The comparative response to individual events between the two probes is also inconsistent; for example, a higher turbidity response is recorded for Event 2 at probe Tb than at probe Ta, despite the increasing distance downstream, while all other events show a higher response at Ta than at Tb. Finally, the relative magnitude of the events also varies e.g. event 3 has the highest recorded response of the five events at T1, but has the second lowest recorded response at Tb. These three indicators of an inconsistent response suggest that the introduced sediment may not be sufficiently

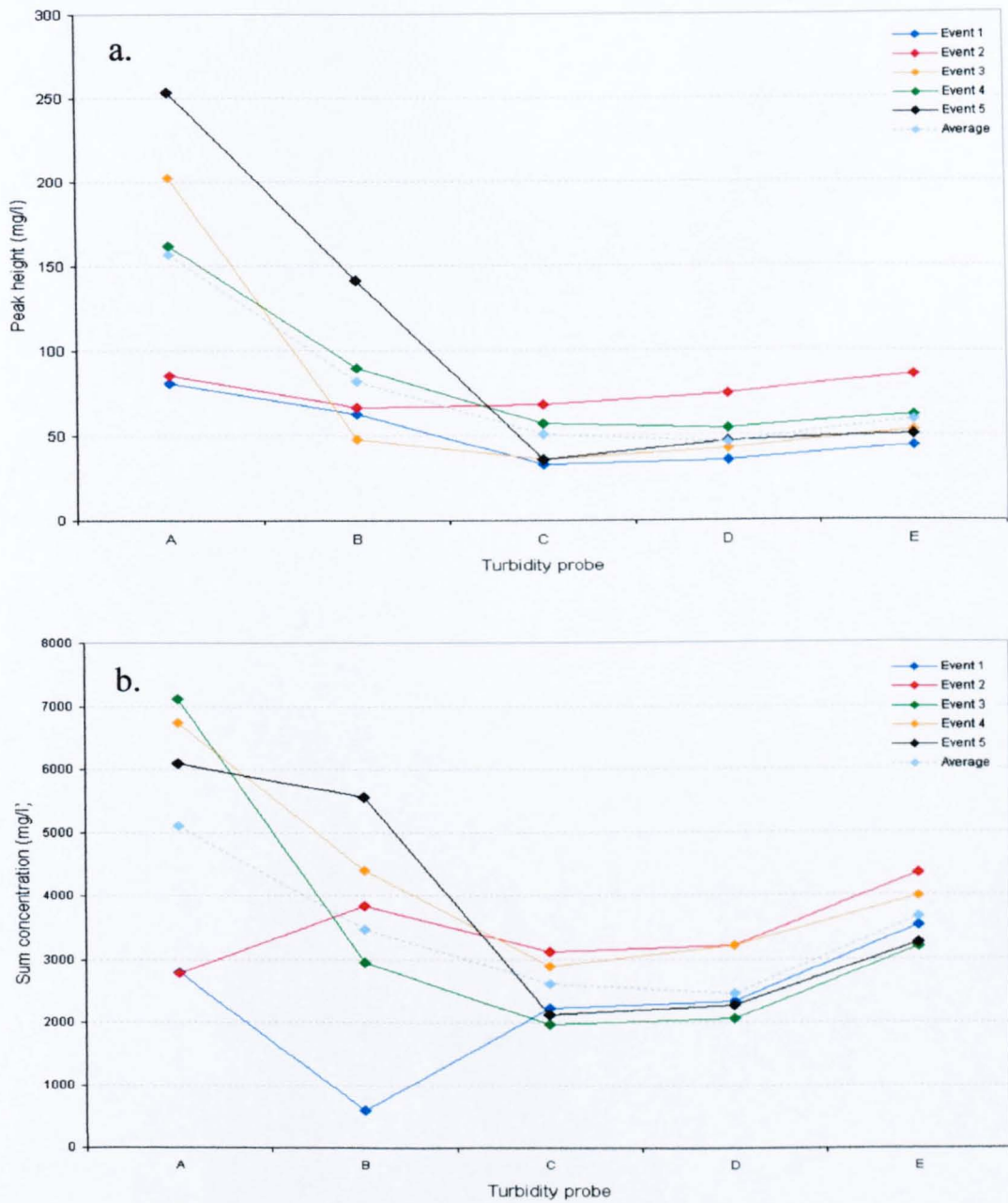


Figure 6.12 Line diagram illustrating the changing magnitude of the introduced sediment concentration for five individual simulated events as recorded by the turbidity probes in the clear water control experiment as expressed by: a) peak concentration; and b) as the sum concentration of the event.

Table 6.3 Descriptive statistics (based on sum concentration) describing the consistency of turbidity probe response to five introduced sediment events during the clear water control experiment. The division between probes of ‘consistent’ and ‘inconsistent’ response describes the approximate ‘mixing distance’ of 1-2 metres: the distance after which sediment is more evenly mixed and probe response is more reliable.

	1. Inconsistent probe response		2. Consistent probe response		
Summary statistics (Sum concentration)	Ta (0 m downstream)	Tb (1 m downstream)	Tc (2 m downstream)	Td (3 m downstream)	Te (4 m downstream)
Mean (mV)	5103.19	3463.01	2460.51	2614.63	3663.35
Maximum (mV)	7120.24	5554.87	3108.25	3212.30	4358.66
Minimum (mV)	2782.55	593.41	1965.32	2057.94	3193.25
Range (mV)	4337.69	4961.45	1142.93	1154.36	1165.41
Standard deviation (mV)	2148.29	1862.51	503.79	550.84	497.96
Coefficient of variation (Cv) (%)	42.10	53.78	20.48	21.07	13.59
Relative magnitude of events	3 > 4 > 5 > 2 > 1	5 > 4 > 2 > 3 > 1	2 > 4 > 1 > 5 > 3	2 > 4 > 1 > 5 > 3	2 > 4 > 1 > 5 > 3

mixed when reaching probes Ta and Tb and that the recorded turbidity responses at these probes are unreliable. Probes Tc – Te, in contrast, display a consistent and reliable turbidity response for each successive event: the recorded magnitudes for each event at each probe are not as wide ranging (standard deviations of 497.96, 503.79 and 550.84 respectively; Cv of 20%, 21% and 14%); the comparative response to each event is the same with a common decrease in magnitude from Tc to Td and an increase in magnitude between Td and Te; and the relative magnitude for each event is also the same from Tc – Te (Table 6.3). The consistent response at probes Tc – Te indicates that sediment mixing is more complete and that the turbidity samples are more representative of the sediment event. Complete mixing is obtained at some point between Tb and Tc and the ‘mixing distance’ may be approximated as 1 - 2 metres downstream of the sediment release point.

Given the suggested approximate mixing length requirement of 1 – 2 m distance from the release point, the probe responses upstream of this may not be representative recordings of the sediment event. In all vegetated experiments, T1 is less than 1 m downstream and the turbidity recordings at these probes must be treated with caution. In the small and medium patch experiments, probes 2, 4 and 5 are positioned less than 2 m but more than 1.5 m downstream from the release point (1.5 m – 1.8 m); these results may also provide inconsistent recordings of events and findings must be considered from this perspective. For the large patch lengths, probes 2, 4 and 5 are positioned more than 2 m downstream and these should provide more consistent responses to events. Similarly, in all cases, T3 is positioned more than 2 m downstream of the release point.

6.6.2 The clear water control: changes in event shape

Figure 6.13 displays both the *magnitude* (sum concentration) and the *shape* of each sediment event at each probe. Figure 6.13a combines the time to peak (T_p) and time to recession (T_r) parameters to give a measure of the symmetry of each peak (perfect symmetry = 1), while Figure 6.13b uses the base width and peak concentration to display a measure of the shape of the peak, i.e. the ratio of peak height to width.

Figure 6.13a demonstrates that the peaks generally have T_p/T_r ratios that place them to the right of the line of symmetry, i.e. the time to peak is longer than the time to recession and this consistent response may largely be an artefact of the method of sediment release. Table 6.4 provides simple summary statistics for the T_p/T_r ratio and supports this visual finding, reporting a mean for all events of 1.38. There are several outliers, however and the T_p/T_r ratio ranges from a maximum of 4.24 to a minimum of 0.22; the standard deviation is also relatively high at 0.86. There are significant differences in the peak symmetry recorded at individual probes but this does not seem to conform to the two groupings established in Section 6.6.1. In general, the three upstream probes, $T_a - T_c$, show more variation in peak shape than the two downstream probes, $T_d - T_e$. T_d and T_e also have very similar mean T_p/T_r values (1.27 and 1.12), both much closer to symmetry than the other upstream probes, and it may be that peak shape becomes more consistent with increasing distance downstream. This would appear logical, as distance and time will inevitably reduce the influence of the initial release characteristics. Further experiments would be needed to confirm this trend.

Figure 6.13b demonstrates that event width remains very similar from probe to probe, despite changes in magnitude. This is well exemplified by Event 1, which has a much greater peak width than the other four events, and this greater width is preserved from probe Ta through to probe Te. This width preservation should also be present in the vegetated experiments unless otherwise influenced by vegetation. Table 6.5 provides summary statistics for base width at each probe and highlights the consistency of base width between probes.

On the basis of the results from the clear water control analysis, base width and sum concentration were chosen as the final descriptive parameters for exploring the vegetated experiments. Sum concentration was chosen, in preference to peak height, as the most representative measure of event magnitude. The investigation of T_p/T_r ratio in the clear water control experiment suggested that peaks may be more consistent in their shape and appear to become closer to a 'mid-peak', symmetrical, shape with increasing distance downstream, but the results were not conclusive and could not be applied with certainty to the vegetated experiments. Instead, it was decided best to concentrate on base width, which was shown to be relatively persistent between probe locations for each event. Any major changes in sum concentration and base width between probes in the vegetated experiments may, on the basis of the control experiment, be reliably ascribed to vegetation influence.

6.7 Approach to the analysis of the vegetated experiments

When the experimental design was first conceived, the intention was that the sediment concentration and peak characteristics recorded at the most upstream probe

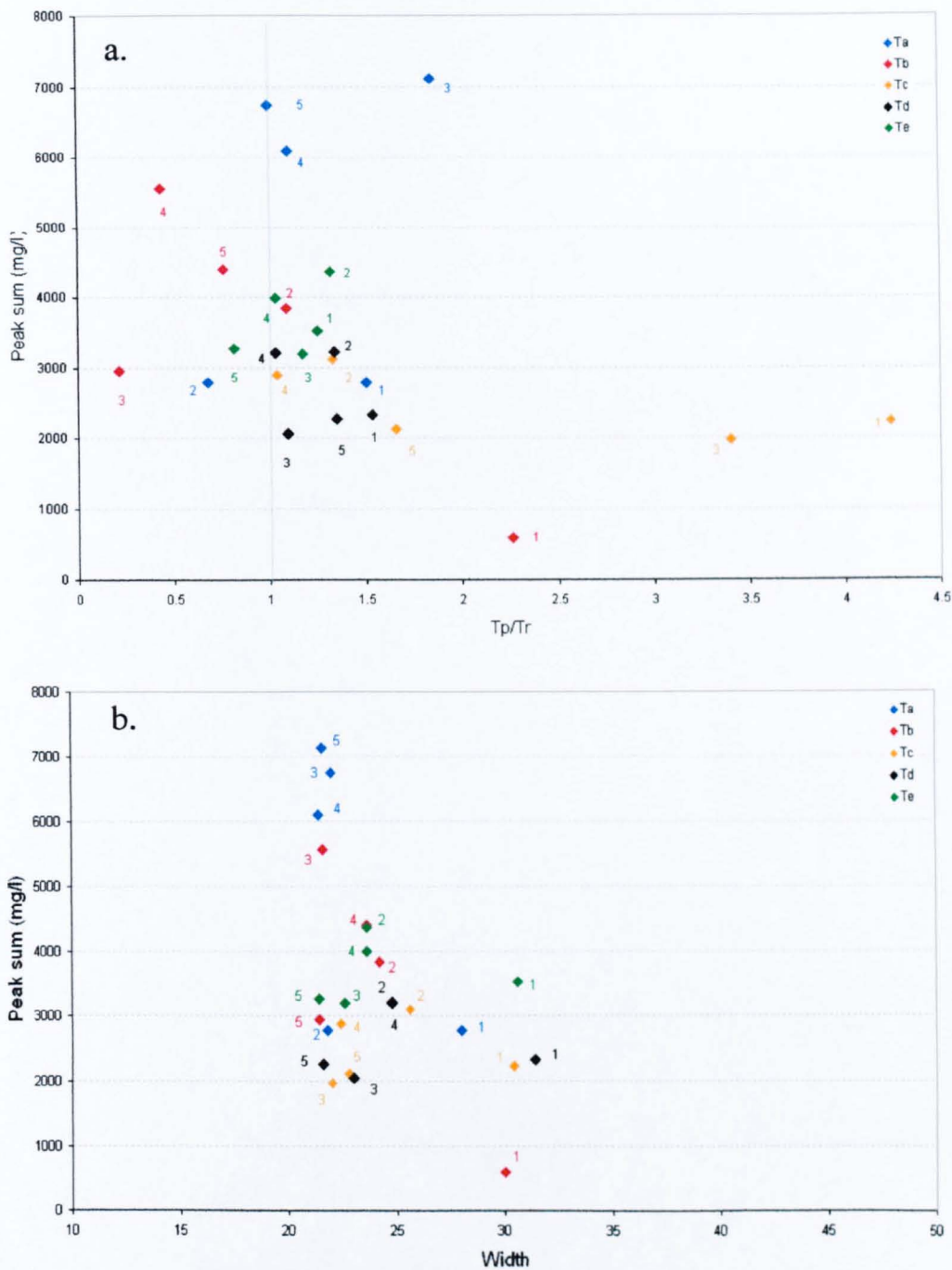


Figure 6.13 Scatter diagrams displaying the *magnitude* (sum concentration) and the *shape* of each sediment event at each probe. **Figure 6.13a** combines the time to peak (T_p) and time to recession (T_r) parameters to give a measure of the symmetry of each peak (perfect symmetry = 1), while **Figure 6.13b** uses the base width and peak concentration to display a measure of peak shape, i.e. the ratio of peak height to width.

Table 6.4 Summary statistics describing the peak symmetry of turbidity probe responses to five introduced sediment events during the clear water control experiment. Symmetry of the peaks is judged by the ratio of time to peak (Tp) and time to recession (Tr); perfect symmetry = 1.

Summary statistics (Tp/Tr ratio)	Ta (0 m downstream)	Tb (1 m downstream)	Tc (2 m downstream)	Td (3 m downstream)	Te (4 m downstream)	All (Ta – Te)
Mean	1.22	0.95	2.33	1.27	1.12	1.38
Maximum	1.84	2.26	4.24	1.53	1.31	4.24
Minimum	0.68	0.22	1.04	1.03	0.81	0.22
Range	1.17	2.05	3.21	0.50	0.50	4.03
Standard deviation	0.45	0.80	1.41	0.21	0.20	0.86

Table 6.5 Summary statistics describing the base width (seconds) of turbidity probe responses to five introduced sediment events during the clear water control experiment.

Summary statistics (Base width)	Ta (0 m downstream)	Tb (1 m downstream)	Tc (2 m downstream)	Td (3 m downstream)	Te (4 m downstream)	All (Ta – Te)
Mean	22.96	24.16	24.64	25.12	24.36	24.25
Maximum	28.00	30.00	30.40	31.40	30.60	31.40
Minimum	21.40	21.40	22.00	21.60	21.40	21.40
Range	6.60	8.60	8.40	9.80	9.20	19.36
Standard deviation	2.83	3.49	3.52	3.76	3.60	3.24

(T1) would be compared to all other downstream probes to quantify the relative change in sediment concentration. However, the clear water control experiment has shown that the sediment concentration, as sampled by the upstream probe, is potentially neither a consistent nor representative estimate of the amount of sediment released. Instead, it is suggested that a mixing distance of 1 – 2 metres exists, after which probes may be considered representative. Therefore, the results of probes 2 – 5 may no longer be compared to T1 but may still be compared relatively to each other, both within and between experiments. The results for T1 are included in each experiment graph for reference but will not be included in the discussion or interpretation of results.

There are ten vegetated experiments, which show a response beyond T1 (Table 6.2); these vary in terms of vegetation size and site location, but may be primarily differentiated by sediment size. Three sediment grades, or size ranges, are represented and Figure 6.14 and 6.15 display the six graphs representing 3 – 4 phi sediment; Figure 6.16 displays the three graphs for 2 – 3 phi sediment; and Figure 6.17 displays the graph for the single 1 – 2 phi experiment. Table 6.6 and 6.7 provide selected summary statistics describing the sum concentration and base width of the sediment events at each probe in each experiment.

6.8 Inside versus outside of the vegetation patch: T2 versus T4 and T5

This section considers the vegetation influence on the three central probes of the probe array (Figure 2.14, Chapter 2) and compares Zones 2b and 2c, as defined in the conceptual diagram (Figure 6.3). Two hypotheses, which consider both event

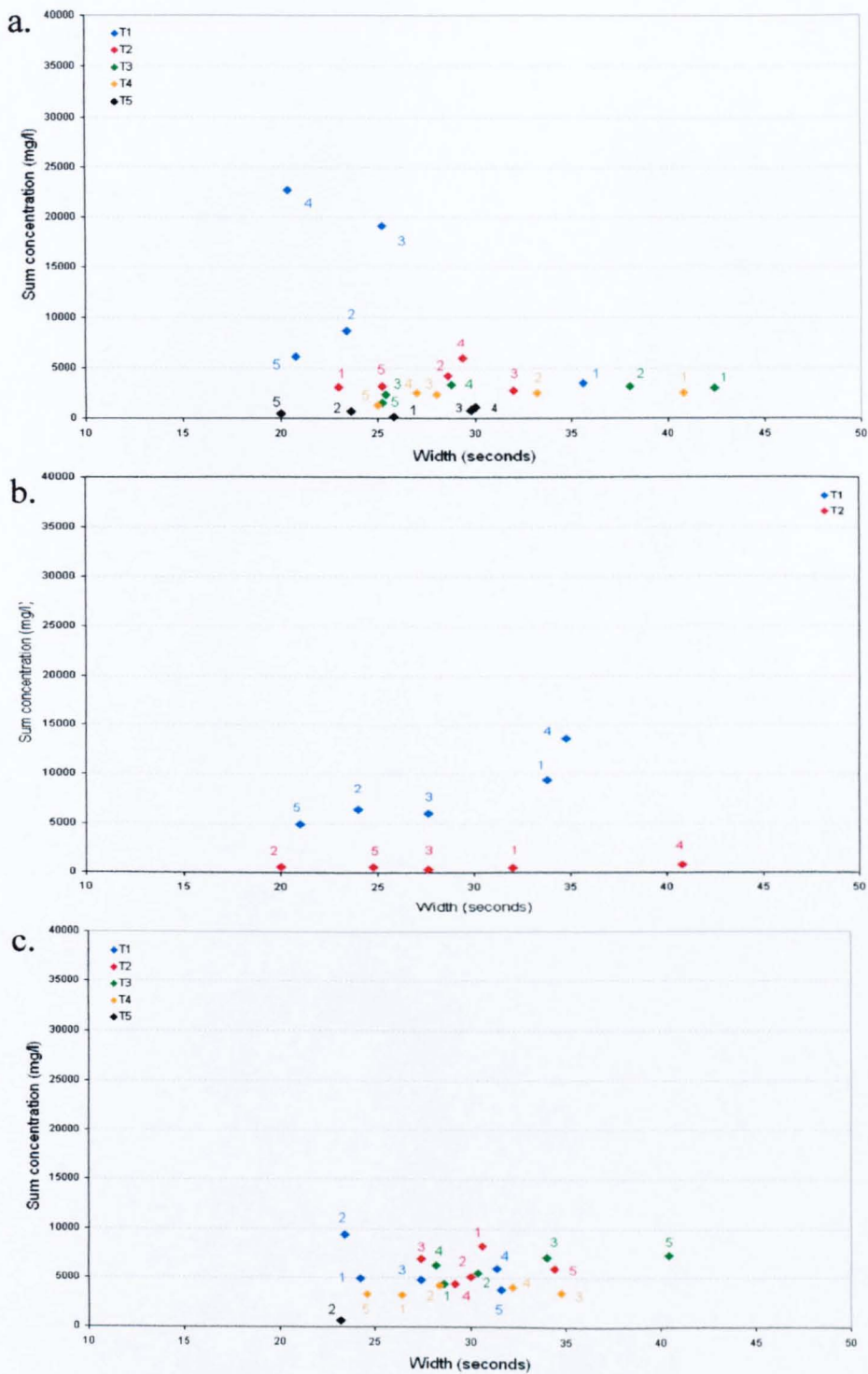


Figure 6.14 Three scatter graphs showing base width and sum concentration for the 3 – 4 phi sediment experiments at Crockways, for: **a)** 'small' *Ranunculus* plant (R1); **b)** 'medium' *Ranunculus* plant (R2); and **c)** 'large' *Ranunculus* plant (R3). Five separate sediment releases were carried out in each experiment and are identified by labels 1 – 5.

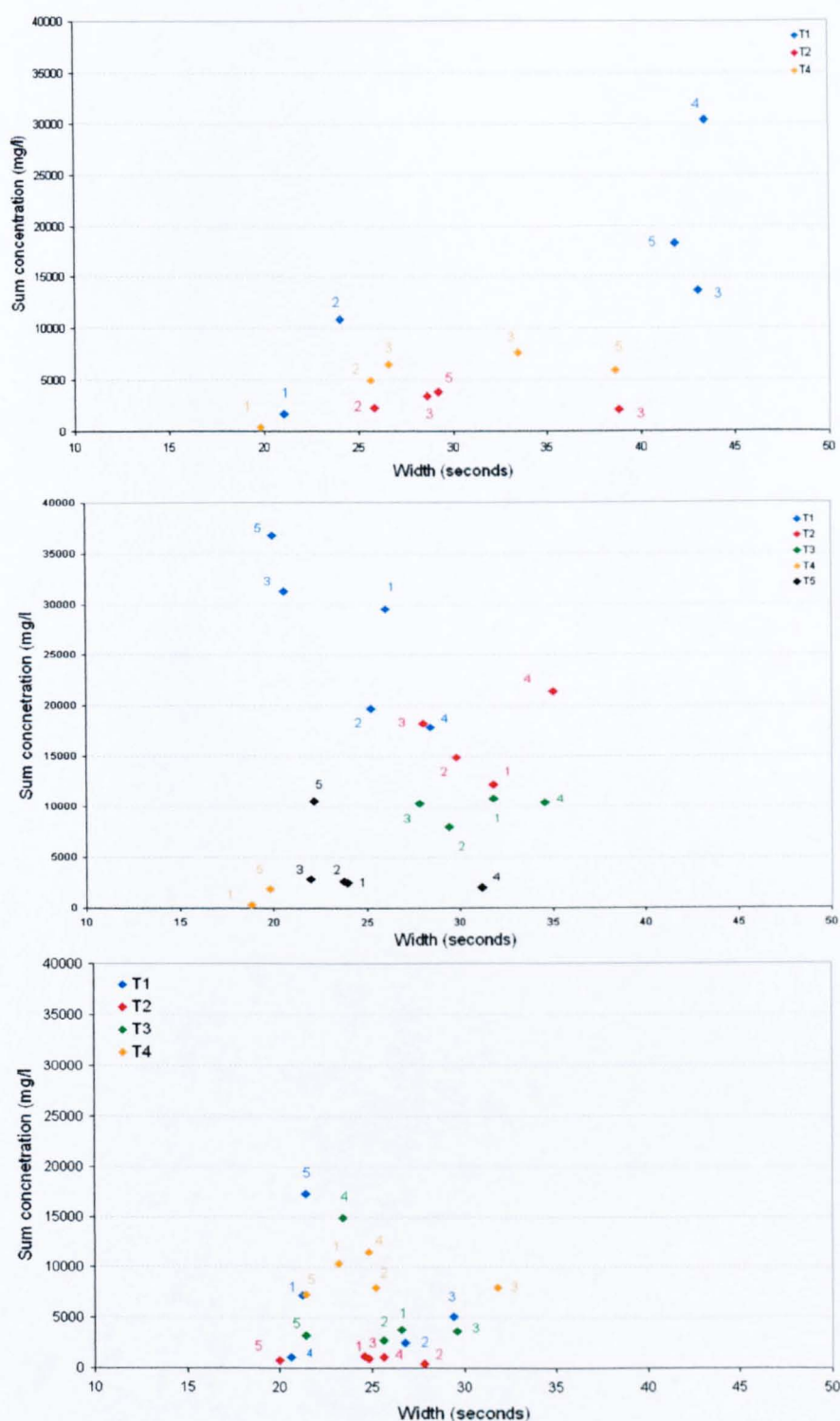


Figure 6.15 Three scatter graphs showing base width and sum concentration for the 3 – 4 phi sediment experiments at Hydeclift, for: **a)** ‘small’ *Ranunculus* plant (R1); **b)** ‘medium’ *Ranunculus* plant (R2); and **c)** ‘large’ *Ranunculus* plant (R3). Five separate sediment releases were carried out in each experiment and are identified by labels 1 - 5.

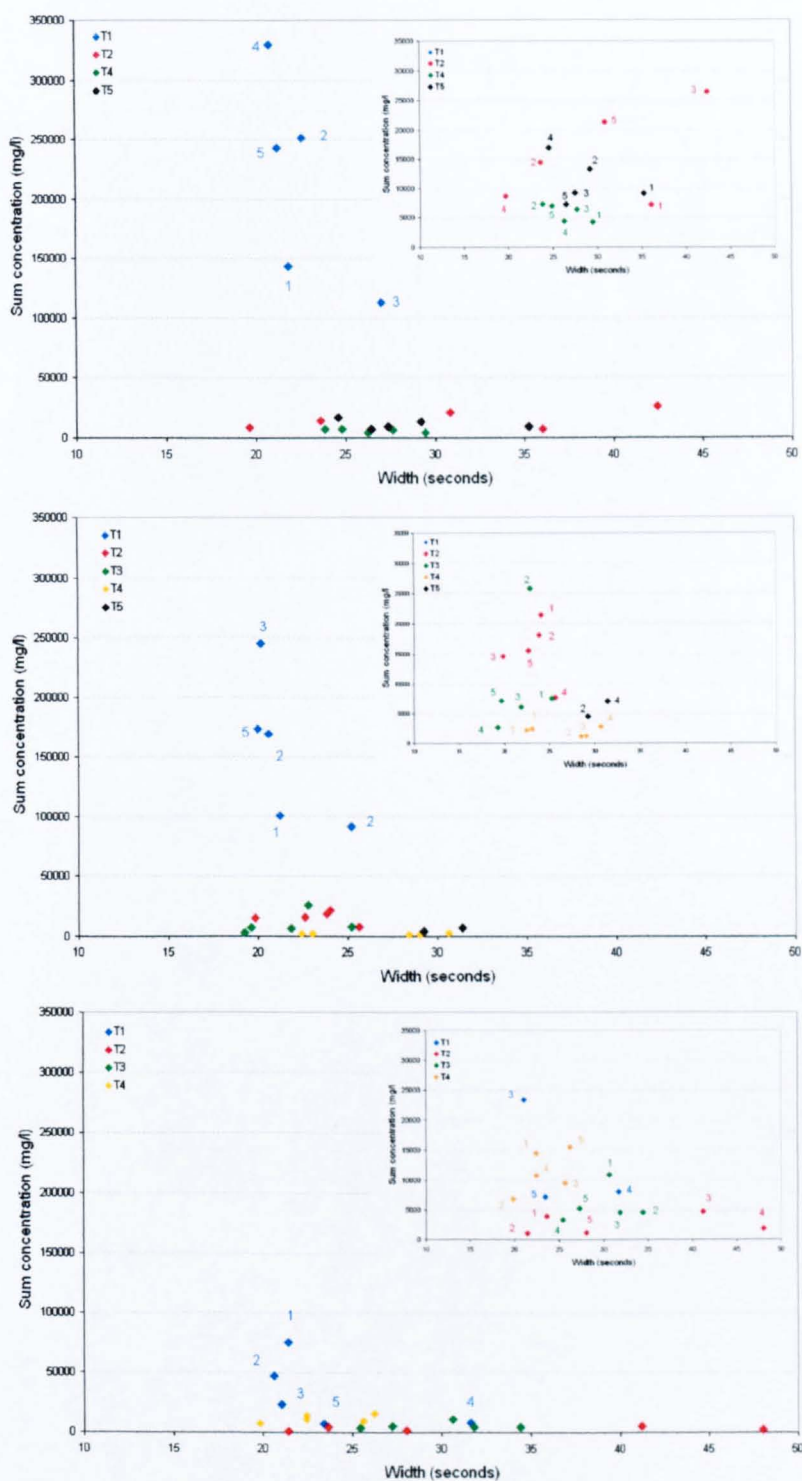


Figure 6.16 Scatter graphs showing base width and sum concentration for the 2 – 3 phi sediment experiments at Hydeclift and Crockways: **a)** ‘small’ *Ranunculus* plant (R1) at Crockways; **b)** ‘large’ *Ranunculus* plant (R3) at Crockways; and **c)** ‘large’ *Ranunculus* plant (R3) at Hydeclift. Five separate sediment releases were carried out in each experiment and are identified by labels 1 - 5.

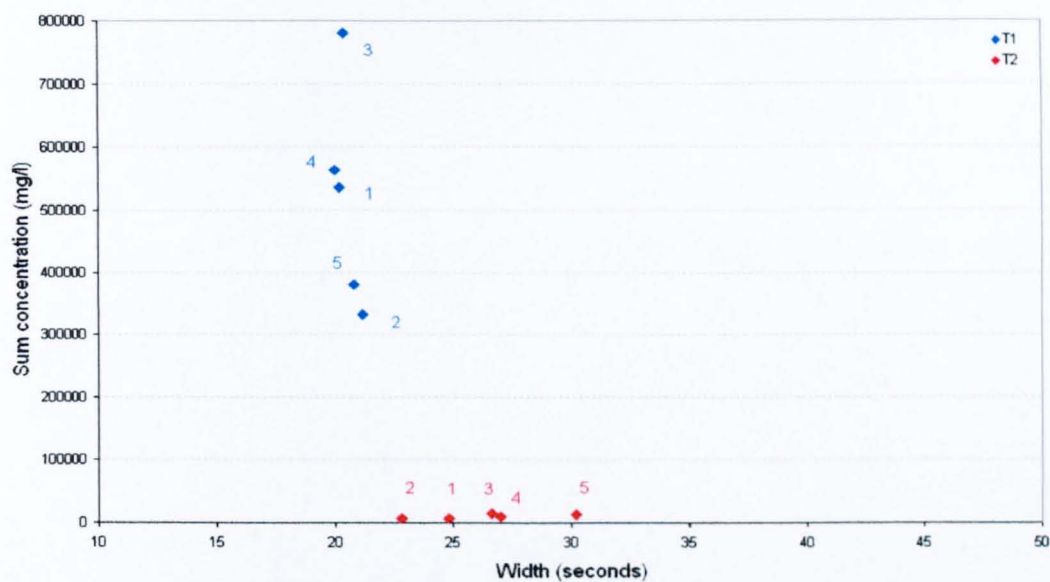


Figure 6.17 Scatter graph showing base width and sum concentration for the 1 – 2 phi sediment experiment for the ‘large’ *Ranunculus* plant (R3) at Crockways. Five separate sediment releases were carried out in the experiment and are identified by labels 1 - 5.

Table 6.6 Selected summary statistics of event magnitude (based on sum concentration) for the 10 turbidity experiments that showed a coherent response to sediment events. The different vegetation patch sizes in each experiment are referred to as R1, R2 and R3; relating to small, medium and large patches *Ranunculus calcareous*.

Experiment	Summary statistics (Sum Concentration)	T1	T2	T3	T4	T5	T2 + T4 + T5
Crockways R3 1 – 2 phi	Mean	518306.42	10166.45	-	-	-	10166.45
	Standard deviation	176785.77	3601.64	-	-	-	-
Crockways R1 2 – 3 phi	Mean	215990.84	15658.72	-	5890.23	11228.95	32777.9
	Standard deviation	87634.41	8239.93	-	1472.77	3893.98	-
Crockways R3 2 – 3 Phi	Mean	155961.00	15482.66	9855.84	1958.50	5771.72	23212.88
	Standard deviation	62817.03	5121.86	9109.26	757.93	1794.83	-
Hydeclift R3 2 – 3 phi	Mean	32217.46	2521.41	5755.11	11404.56	-	13925.97
	Standard deviation	28931.96	1690.38	2970.20	3573.67	-	-
Crockways R1 3 – 4 phi	Mean	12022.53	3798.22	2665.63	2202.14	660.01	6660.37
	Standard deviation	8432.52	1328.60	714.87	531.75	344.08	-
Crockways R2 3 – 4 phi	Mean	8050.04	527.68	-	-	-	527.68
	Standard deviation	3517.66	175.30	-	-	-	-
Crockways R3 3 – 4 phi	Mean	5757.70	6092.82	6088.42	3622.34	615.28	10330.44
	Standard deviation	2179.14	1511.69	1231.93	439.36	-	-
Hydeclift R1 3 – 4 phi	Mean	14136.94	2571.39	-	4855.65	-	7427.04
	Standard deviation	10520.05	847.46	-	2765.03	-	-
Hydeclift R2 3 – 4 phi	Mean	63300.57	16628.69	9853.71	306.31	2472.49	19407.49
	Standard deviation	8036.64	3989.69	1265.95	1073.78	3606.82	-
Hydeclift R3 3 – 4 phi	Mean	6472.57	857.51	6237.86	9414.47	-	10271.98
	Standard deviation	6426.35	298.27	5202.83	1820.17	-	-

Table 6.7 Selected summary statistics (for base width) for the 10 turbidity experiments that showed a coherent response to sediment events.

Experiment	Summary statistics (Base Width)	T1	T2	T3	T4	T5
Crockways R3 1 – 2 phi	Mean	20.52	26.28	-	-	-
	Standard deviation	0.48	2.75	-	-	-
Crockways R1 2 – 3 phi	Mean	22.68	30.48	-	26.36	28.56
	Standard deviation	2.51	9.20	-	2.22	4.07
Crockways R3 2 – 3 Phi	Mean	21.44	23.16	21.72	26.68	30.3
	Standard deviation	2.15	2.16	2.46	3.73	4.69
Hydeclift R3 2 – 3 phi	Mean	23.6	32.44	29.88	23.28	-
	Standard deviation	4.60	11.60	3.60	2.63	-
Crockways R1 3 – 4 phi	Mean	25.08	27.64	31.96	30.80	25.84
	Standard deviation	6.20	3.55	7.81	6.36	4.25
Crockways R2 3 – 4 phi	Mean	28.24	29.04	-	-	-
	Standard deviation	6.02	7.89	-	-	-
Crockways R3 3 – 4 phi	Mean	27.60	30.32	32.32	29.28	23.20
	Standard deviation	3.86	2.58	5.06	4.18	-
Hydeclift R1 3 – 4 phi	Mean	34.64	30.60	-	28.80	-
	Standard deviation	11.15	5.66	-	7.30	-
Hydeclift R2 3 – 4 phi	Mean	24.04	31.15	30.90	19.30	24.64
	Standard deviation	3.62	3.00	2.96	0.71	3.78
Hydeclift R3 3 – 4 phi	Mean	23.88	24.56	25.32	25.28	-
	Standard deviation	3.97	2.85	3.13	3.94	-

magnitude and event shape, are proposed to help structure the analysis: (1) that the proportion of sediment travelling inside (T2) and outside (T4 and T5) the vegetation will vary according to vegetation patch size and sediment type; and (2) that the base width of the sediment events will be larger inside the vegetation than outside the vegetation due to velocity differences and that this may also vary with patch size.

For the first hypothesis, it was envisaged that, in the absence of vegetation, the sum concentration of the sediment events would be higher at T2 than at T4 and T5, because T2 is directly in line with the sediment source point. However, this scenario may not be replicated in the presence of vegetation: attenuation of sediment by the vegetation may greatly *reduce* the sediment signal before it reaches T2, while the velocity 'dead zone' imposed by the vegetation may actively divert water flow and suspended sediment around the vegetation and *increase* the concentration observed at T4 and T5. Depending on the strength of these combined effects, it is even possible that no sediment signal will be observed at T2. The attenuation and diversion effects are likely to be most influential at larger patch sizes, where the vegetation dead zone is more pronounced, than at small patches, and the following analysis considers changes to the dominant sediment flow path dependent on patch size, sediment grade and site conditions. The second hypothesis was based on the knowledge that water velocity within the vegetation is lower than outside the vegetation. Sediment events are likely to pass more slowly, resulting in a longer base width, inside the vegetation, while the higher velocities outside of the vegetation would suggest that the event will pass more quickly resulting in a shorter base width.

6.8.1 Sum concentration: 3 – 4 phi experiments

There are six experiments for the finest sediment grade (3 – 4 phi): three at Crockways (Figure 6.14) and three at Hydeclift (Figure 6.15). At Crockways, the sediment signal inside the vegetation is consistently higher than that outside the vegetation for each patch length. Internal transport therefore represents the dominant transport route (see Table 6.6) and this indicates that the effects of the vegetation at Crockways are not sufficient to alter the transport pattern from what would be expected in the absence of vegetation. In the small patch experiment, the event-averaged sum concentration at T2 (3798.22 mg/l) is appreciably higher than for T4 or T5 (56% and 17% of the T2 total respectively), and a similar pattern may be observed in the large vegetation patch, the sum concentration at T2 (6092.82 mg/l) is again much higher than for T4 (59 %) and T5 (10 %). In the medium patch experiment, overall sediment attenuation is much higher than for the other two experiments; the sediment signal is only weakly evident at T2 (527.68 mg/l) and is not evident at either T4 or T5, but this still reveals that internal sediment transport is dominant.

The results from Hydeclift are not so consistent. In the small and large patch experiments, the amount of sediment travelling outside (T4 and T5) the vegetation is greater than that travelling within, while in the medium patch the opposite situation is found: the sum concentration recorded at T2 inside the vegetation, is greater than T4 and T5 outside. In the small patch experiment, while there was no response evident at T5, the average concentration at T2 (2571.39 mg/l) is considerably lower than at T4 (189 % of the T2 total). At the large patch experiment the difference is

even greater; at T2 the concentration is very low at only 875.51 mg/l, and though a sediment signal is not evident at T5, it is very high at T4 (1098 % of the T2 total). These two experiments would seem to suggest that either sediment attenuation is greater in the larger patch and/or the diversion of water flow and sediment around the patch is much greater. Either way, the findings suggest that the large patch has a greater influence on passage of sediment than the small patch. However, the medium patch does not follow this pattern: the sum sediment concentration at T2 (16628.69 mg/l) is significantly higher than at T4 and T5 (9853.71 mg/l and 306.31 mg/l respectively) and the changes in sediment proportions at Hydeclift cannot be said to follow the patch size hypothesis. The upstream velocity at the medium patch is significantly lower than for the other two patches, due to the influence of upstream vegetation, and it is likely that this has controlled the dominant sediment transport route. The lower velocity at the medium patch allows water to pass through the vegetation and not be diverted quickly around it. The fact that the concentration is lower inside the vegetation than outside in the small and large patch experiments, suggests that vegetation has some effect on sediment concentration and that this effect may be greater in the shallow depth environment at Hydeclift than at the deeper Crockways site.

Site differences may be best displayed by directly comparing the combined sum concentrations from each experiment. The average sum concentrations at T2, T3 and T4 were added together to give an indicator of how much of the original sediment load reached the half-way distance mark at each vegetation stand and are shown in Table 6.6. These amalgamated values were compared between patch sizes and site locations. At Crockways, the attenuation between the release point and T2 is,

surprisingly, least in the large patch experiment and highest in the medium patch. This may suggest that factors other than patch length have influenced the results and that trapping efficiency is not just a simple expression of patch size. At Hydeclift, the attenuation values follow the expected pattern, with attenuation lowest at the small patch, and highest at the large patch and the medium patch intermediate in its attenuation. The medium patch attenuation, however, is much closer to the large patch attenuation value than the small patch value and this shows a large increase in attenuation between the two smaller patch sizes, suggesting that attenuation is not a linear function of patch length.

The amalgamated values complicate the influence of vegetation patch size in that two counterbalancing effects are operating concurrently: (i) the large patches are expected to have a greater attenuation effect than smaller patches in that a longer filter length will remove more sediment; thus lowering the amalgamated value, but equally (ii) larger patches are predicted to cause greater diversion of flow and sediment around the plant, which will increase the amalgamated value. It is the balance between the two effects of attenuation and diversion that will determine the overall amalgamated value and this balance may differ greatly between individual plant stands.

The results from all six experiments show that the 3 – 4 phi sediment does pass through the vegetation and that the vegetative filter is not total. The results also indicate that the filtering effect is variable and is dependent on vegetation length, water depth and water velocity.

6.8.2 Sum concentration: 2 – 3 and 1 – 2 phi experiments

The 2 – 3 phi experiments at Crockways (Figure 6.16a and 6.16b) generally support the findings of the finer 3 – 4 phi experiments, in that the sediment concentration outside the vegetation is again lower than that observed within the vegetation. In the small patch experiment, the sum concentration at T2 (15658.72 mg/l) is higher than both T4 (38 % of the T2 total) and T5 (72 %), while in the larger patch this response is more extreme: T2 has an average sum concentration of 15482.66 mg/l while T4 and T5 are only 13 % and 37 % percent of the T2 total. The medium vegetation experiment has the highest overall sediment attenuation, with no sediment signal evident in any of the downstream probes (Table 6.2). These results indicate that the overall pattern of vegetation induced changes in sediment, i.e. inside versus outside the patch, is consistent between sediment types at Crockways and is not sediment size-specific.

There is only one 2 – 3 phi experiment at Hydeclift that registers a response downstream of T1, and this occurs at the largest vegetation patch (Figure 4.14c). The pattern of sediment transport is consistent with that from the 3 - 4 phi experiments and the dominant sediment pathway is unchanged; i.e. external transport (11404.56 mg/l at T4) is greater than internal transport (2521.41 mg/l at T2). However, the overall level of attenuation is not consistent with the 3 – 4 phi experiments; the large patch displayed the highest attenuation of the finer sediment (Table 6.6), yet is the only experiment represented at the 2 – 3 phi sediment grade. This suggests that though the overall balance between attenuation and diversion

processes remains the same, that the filter strength of the plant at Hydeclift may differ depending on the sediment size considered.

There is only one 1 – 2 phi experiment, which shows a demonstrable response downstream of T1, and this is at the large patch experiment at Crockways (Figure 6.17). This again suggests consistency in response between sediment sizes at Crockways as the large patch showed the least attenuation in the 3 – 4 phi experiments. The experiment for 1 – 2 phi shows a response only at T2 and this suggests that more sediment travels through, rather than around the vegetation and is consistent with the pattern observed in both the 3 - 4 phi and 2 – 3 phi experiments.

The attenuation of the 2 – 3 phi sediment experiments may also be examined by combining the average sediment values recorded for T2, T3 and T4 (Table 6.6). However, consideration of the 1 – 2 phi attenuation is of little value without other experiments to compare it to. When the combined concentrations for the 2 – 3 phi experiments at Crockways are compared, very similar results are found to those using the finer grade sediment. The small patch shows least attenuation, while the medium patch, which showed most attenuation at the finer grade, shows total attenuation. The large patch attenuation is less than that of the small patch. At Hydeclift, only one patch is represented in the 2-3 experiments, and this is in opposition to the finer grade experiments; the patch with the highest attenuation at the finer grade is the only patch where attenuation is not complete at the coarser 2 – 3 phi grade. For the single 1 – 2 experiment at Crockways, only the large patch experiment is represented downstream and this is the patch that showed least attenuation at the 2 – 3 and 3 – 4 phi grades.

The results at the coarser 2 – 3 phi and 1 – 2 phi sediment grades reveal that the overall pattern of vegetation induced changes in sediment, i.e. the proportion of sediment travelling inside versus outside the patch, is consistent between sediment types at both sites. However the pattern of overall attenuation differs between sites: at Crockways the attenuation follows the attenuation pattern found at the finer 3 – 4 phi grade, while at Hydeclift the effects differ. This site difference highlights the complicated nature of macrophyte-sediment interactions.

6.8.3 Base width: 3 – 4 phi; 2 – 3 phi and 1 – 2 phi experiments

The average base widths for all events and all ten experiments are shown in Table 6.7 and are visually displayed in Figure 6.14 – Figure 6.17. The overall average base width at T2 is 28.57 seconds compared to 26.22 and 26.51 seconds at T4 and T5 and the results seem to support the theory that base width is likely to be longer inside vegetation than outside. However, this average result hides differences in the results from individual experiments. Of the ten experiments considered, only eight show a response at T2 and at either T4 or T5; in six of these the sediment signal is longer inside the vegetation while at two of these the event base width is longer inside rather than outside the vegetation. Thus, it seems that the sediment travel time may not always be affected by the characteristics of the individual vegetation stand and that other factors, e.g. other vegetation stands in close proximity, may have had some influence. Base width may also be linked to the magnitude of an event, with a larger event having a longer base width and *vice versa*, meaning that a reduction in the

sediment sum concentration may also affect the event base width. There is little evidence that travel time changes according to patch length.

The two hypotheses presented in this section (hypotheses 1 and 2) appear to be applicable under certain conditions, but, in general, they are too simplistic to cover the complex interactions that govern sediment capture by macrophytes. The hypotheses need to take into account factors other than sediment grade and vegetation length. Possible contributory factors identified in this section include: water depth; water velocity and the proximity of other vegetation stands.

6.9 Downstream attenuation of sediment: T2 versus T3

This analysis section concentrates on Zone 3 from the conceptual diagram (Figure 6.3) and relates the observed characteristics in Zone 3 to the analysis of Zones 2b and 2c above. As with the previous section, two hypotheses will be used to structure the analysis: (3) that attenuation of the sediment signal between T2 and T3 is likely to vary according to patch size and sediment size, with greater attenuation likely in the larger plant stands and with the coarser sediment grades; and (4) that the recombination of the separate sediment signals from T2, T4 and T5 will vary according to patch size, and that recombination is more likely after small patches than after large, because the individual sediment signals will have been separated for less time and over a shorter distance.

6.9.1 Downstream attenuation: 3 – 4 phi experiments

The majority of the 3 - 4 phi experiments show an attenuation of sediment concentration between T2 and T3 but the degree of attenuation does not seem to follow that outlined in Hypothesis 3 (Table 6.6). At Crockways, in the small vegetation experiment, attenuation from T2 to T3 is around 30 %, but at the large experiment attenuation is only very slight, with the sediment concentration recorded at T3 only 0.1% less than at T2. The largest attenuation occurs at the medium vegetation where no sediment response is evident at T3, though this is not surprising given the very low response at T2. At Hydeclift, the greatest attenuation is at the small vegetation experiment (100%), while the medium experiment demonstrates an attenuation of 41 %. The large experiment meanwhile, confounds theory entirely and shows a much *larger* sum concentration at T3 than at T2, an increase in concentration of 727 %. These experiments, particularly the large experiment at Hydeclift, demonstrate that T3 not only records attenuation of sediment by the vegetation but also describes the recombination of sediment after the vegetation patch. T3 is located immediately downstream of the vegetation patch and this indicates that recombination occurs very quickly after the patch and perhaps even occurs before the downstream end of the vegetation stand. The sediment concentration at T2 cannot be meaningfully related to T3 because there is no way of separating the two processes of sediment attenuation and recombination. A separate turbidity probe placed just upstream of T3 and still within the vegetation would have allowed the processes of attenuation and recombination to be differentiated and this modification should be included in any future experiments.

T3 turbidity readings are still valuable however, and a comparison of T3 readings from different experiments provides a measure of *overall* sediment attenuation i.e. from the point of release upstream of the vegetation to downstream of the vegetation at T3. Overall, attenuation appears to be greatest for small and medium vegetation stands and least for large vegetation stands (Table 6.7). This is in opposition to hypothesis 3, but may be explained by the increased diversion of flow and sediment around, rather than through, the larger vegetation stands due to the greater velocity barrier (Section 6.8.3); smaller vegetation stands allow water to pass through their canopy where sediment may be attenuated. The results are also in opposition to hypothesis 4 and show that recombination is greatest in the larger plant stands, largely because more sediment travels around the large vegetation stands and there is less attenuation.

6.9.2 Downstream attenuation: 2 – 3 phi and 1 – 2 phi experiments

The 2 – 3 phi experiments reflect the results shown in the 3 – 4 phi experiments. At Crockways attenuation between T2 and T3 is 100 %, while at the large patch attenuation is only 36 %. At Hydeclift the recombination of sediment pathways after the large patch is again evident, with the sum concentration at T3 (5755.11 mg/l) representing a 228 % increase from the T2 concentration. For the single 1 – 2 phi sediment, sediment attenuation between T2 and T3 is 100 %. These results are again in opposition to hypotheses 3 and 4 but may be explained by the dominance of diversionary transport in the larger patches.

The location of the turbidity probes (T2 and T3) means that attenuation and recombination cannot be separated and this means the hypotheses in this section (3 and 4) are harder to assess. However, both processes may be assessed together, and the results indicate that downstream sediment attenuation is least (and recombination greatest) at the larger vegetation patches, while at the small and medium plant stands sediment attenuation is greatest (and recombination least), due to the diversion of greater sediment volumes around the plant at larger plant stands. The analysis also reveals that recombination occurs at a very short distance downstream, and perhaps even occurs in the free space under the trailing vegetation, before the downstream end of the plant.

6.10 Discussion

This chapter has sought to evaluate previous research into macrophyte-sediment interactions and to deploy an experimental methodology which would help to fill gaps in current knowledge. In the following discussion, the results and analysis presented in the chapter are integrated to assess their contribution to existing knowledge and to evaluate the success of the experimental method.

6.10.1 Questions revisited

In Section 6.1 several linked questions were put forward to highlight the gaps in the current knowledge, these are revisited here to review the findings from the microscale experiments. Three questions were considered:

- i. What are the changes in suspended sediment concentration before flow enters and after flow exits vegetation stands? What changes occur within the vegetation? How does this compare to what happens outside the vegetation in the ambient flow stream?

The analysis in this chapter set out to examine the changes in sediment concentration which occur in the time and space between water entering and exiting the vegetation. The 'washing' experiments indicated that sediment is trapped by the vegetation and that sediment concentration will be reduced downstream when compared to the upstream initial conditions. The washing experiments also revealed that the volume of trapped sediment is not uniform along the length of the plant stand, and that sediment concentration is likely to reduce progressively as the water travels through the vegetation. The turbidity experiments were designed to quantify these change in sediment concentration but, due to methodological problems, absolute comparisons of initial (T1) and final (T3) sediment concentrations could not be made and less was revealed about overall attenuation values than was originally hoped. Some quantitative comparison was possible between the middle of the vegetation patch (T2) and downstream of the patch (T3), and the majority of experiments showed a decrease in sediment concentration between T2 and T3; suggesting that sediment concentration *is* reduced between upstream and downstream of the vegetation. However, the experiments revealed that T3 does not just record attenuation of sediment by the vegetation, but describes the recombination of sediment travelling both inside and outside the vegetation patch. The sediment concentration at T2 cannot be reliably compared to T3 because there is no way of separating the two processes of sediment attenuation and recombination. The results at T3 do have

some value when considered alone. A comparison of T3 values between experiments also indicates that the filtering effect of the vegetation is variable and is dependent on sediment grade, vegetation length, water depth and water velocity. The T3 values also show that some fine sediment does pass through the vegetation and that the vegetative filter is not total.

The turbidity experiments reveal more about the difference in sediment concentration between outside and inside the vegetation patch than they do about downstream and upstream. The experiments revealed the existence of two counterbalancing processes which operate concurrently: (i) *attenuation* of sediment as it travels through the vegetation; and (ii) *diversion* of sediment around and above the vegetation caused by the velocity 'dead zone' within the vegetation. For some patches internal transport represented the dominant transport route, and for others external transport is dominant, and it is the balance between the two effects of attenuation and diversion that will determine the overall sediment concentration downstream of the vegetation. This balance may differ greatly between individual plant stands depending upon vegetation length and site conditions, especially initial water velocity. The sediment concentrations recorded at T3 indicate that the recombination of sediment travelling outside and inside the vegetation occurs very quickly after the vegetation patch and may even occur before the downstream end of the vegetation stand.

Sediment concentration in the ambient flow stream outside the vegetation is likely to be less than the initial conditions due to dispersion effects, but in certain circumstances the concentration may be greater than if vegetation were not present: e.g. the barrier presented by two adjacent stands of vegetation may serve to limit

dispersion and concentrate the sediment in a preferential flow channel. The present experiments could not distinguish any changes in the sediment concentration that may have occurred outside the vegetation due to erosion of the stream bed in the fast flow channels.

- ii. Is sediment retention within macrophytes size selective? What is the range of particle sizes retained by the plant and which sizes are not retained? Do both suspended sediment and bedload contribute to deposition within macrophytes?

The investigations in this chapter have found that sediment retention within macrophytes is size selective. The ‘washing’ experiments show quantified values for the sediment sizes retained, which change depending on the architecture of the plant and the shape of the plant leaves, the position of the plant in the channel and antecedent discharge conditions and the mode of sediment deposition. For fine-leaved submerged plants such as *Myriophyllum alterniflorum*, and analogously *Ranunculus calcareous*, a downstream progression in sediment trapping characteristics are also observed e.g. more large particles were trapped in the central sections of the plant (*Myriophyllum* 2 and 3) and more small particles were found in the most downstream portion of the plant (*Myriophyllum* 4). For the *Myriophyllum* plant as a whole, three overlapping sediment size distributions were observed, identified by the mode of their peaks, at: (i) ~ 20 μm ; (ii) ~ 160 μm and (iii) ~ 950 μm .

Patterns based on sediment size were also evident in the turbidity experiments, sediment. In the peak ‘presence and absence’ testing (Section 6.5.2) there is a

consistent difference in the number of experiments showing a response downstream of T1; the larger sediment grades showed less response downstream than the smaller particles and this suggests that the larger sediment particles are retained in greater volume than the smaller particles. Similarly, in the analysis of sediment sum concentrations (Section 6.8 and Section 6.9), the overall attenuation of sediment was greatest for the larger sediment grades and least for the finer sediment grades. All sediment grades showed some attenuation of sediment meaning that a range of particle sizes are retained but in varying amounts.

When considered in tandem, the washing and turbidity experiments would seem to suggest that both suspended sediment and bedload make a contribution and that contributions vary with discharge. Coarser sediment grades were present in the vegetation washings, even in sub-sample 4 of the *Myriophyllum* plant, while similar sediment sizes introduced to the flow in the turbidity experiments produced little response downstream of T1. This discrepancy suggests one of three things: (i) that there are significant differences in the trapping efficiency of *Myriophyllum* and *Ranunculus* plants; (ii) that coarser sediments enter the plant canopy predominantly as bed load; or (iii) that coarser sediments enter the plant canopy as suspended sediment but under higher discharges than that experienced during the July turbidity experiments.

In the ambient flow stream, alongside and above the vegetation, D50 (median sediment size) is likely to remain similar to the upstream conditions. The results in Section 6.8 showed that, overall, the degree of attenuation of sediment from upstream to downstream changed with sediment type but the 'pattern' of sediment

transport (internal versus external transport) did not. Coarser sediments followed the same pattern as finer sediment and there does not seem to be a sediment size that is more likely to travel through the vegetation. Downstream of the vegetation, in the recombination zone (Zone 3), D50 is likely to be lower than the initial conditions because coarser sediment experiments showed higher attenuation in the turbidity experiments. The present experiments could not distinguish any changes in the sediment population that may have occurred outside the vegetation due to erosion of the stream bed in the fast flow channels.

iii. How do macrophyte-sediment interactions change with differences in vegetation characteristics, water velocity and water depth?

Vegetation length does not appear to be a definitive measure of vegetation characteristics and attenuation does not become greater as vegetation size increases. Other factors complicate the relationship, for example differences in vegetation density, differences in initial velocity conditions, and the effects of water depth on vegetation density. The restriction in stand height imposed by the shallow water at Hydeclift means the plants may grow more thickly and with fewer gaps beneath the vegetation. In the deeper water at Crockways, the terminal parts of large *Ranunculus* stands tend to float at or just below the water surface, and the vegetation density at lower depths may be less than in smaller stands and in shallower environments. If the sediment travels under the vegetation then sedimentation is likely to be less.

The balance between sediment attenuation and diversion is very important (see Section 6.9). Large plants were predicted to have a greater capacity to attenuate

sediment due to the greater velocity reduced and larger storage area. However, the greater velocity reduction creates a stronger barrier to flow and sediment transport and more flow is diverted around the plant and attenuation appears to be *less* in larger plants. This balance may shift at a critical mass, e.g. the relationship between attenuation and plant size in Section 6.9 appears to be non-linear, with greater change apparent between medium and large vegetation than between small and medium vegetation. The balance in attenuation and diversion is also likely to change between plant stands of similar vegetation characteristics but with different initial water velocities and at the same vegetation stand with changes in discharge (and hence velocity).

The experiments at the microscale have served to show the complexity of the macrophyte filter; each vegetation patch is a unique filter, with variable characteristics which change over the growing season and which may vary throughout the length of the patch. Each individual filter will interact differently with water flow and with sediment depending on: patch length, patch width and density; water velocity; water depth; sediment type and the proximity of neighbouring vegetation. Overall, it appears that to compare macrophytes to a filter is perhaps too clinical a description of their effects on sediments; suggesting as it does a standardised process with a predictable output.

6.10.2 Additional sources of complexity

The above analyses have revealed that the measured sediment characteristics do not appear to change in a systematic manner with changes in vegetation patch length.

This suggests that the basic ‘small’, ‘medium’ and ‘large’ designation of the plant patches may be too simplistic and that other factors may determine the strength of the vegetative effects. Vegetation length has been deemed the most important factor in studies looking at the effectiveness of vegetated filter strips in waste treatment but this may not translate so well in the river context given the limited width of the vegetation stands compared to reed bed filters. Thornton *et al.* (1997) showed the complexity of the vegetative filter when they compared the sedimentation potential of four species of river vegetation. They took account of such factors as the cross-sectional area of the vegetative stem, the circumference and length of the stem, and the density of the vegetation. The plant stands used in the present study are all from the same species, *Ranunculus calcareous*, but each stand may vary, not only in length, but also in width, height, density and maturity and may consist of one large or several smaller entwined plants. Each of these factors will have some effect on the filter characteristics and will complicate macrophyte-sediment interactions such that a simplistic, size-based classification of the plant stands is not sufficient. Another source of complexity is the interaction with neighbouring plants. For example, Table 6.2 details the velocity conditions upstream of each site and showed that initial velocities varied greatly between experiments, from 21.95 m s⁻¹ to 91.98 m s⁻¹. These velocity differences are conditioned by vegetation density and pattern (Chapter 4) and partly reflect the position of the plant stand within the channel but are also greatly conditioned by upstream vegetation. It would have been preferable to have comparable initial conditions for all the sample stands but this was difficult to achieve without removing adjacent vegetation.

One final variable not considered is the amount of sediment trapped in the plants before the experiments. If a plant canopy were close to its theorised 'storage capacity' then less sediment might be captured during the turbidity experiments than if little sediment was stored in the plant prior to the experiments. A quick comparison of the sediment storage beneath the vegetation patches was gained by measuring sediment depth at 0.2 m intervals along the central longitudinal axis of each plant stand, the results of which are summarised in Table 6.8. If sediment storage were influential then it might be expected that the turbidity experiments showing least attenuation might be matched by high prior sediment storage, but instead the opposite appears true. For example, the highest sediment attenuation at Crockways is at the medium patch and this is matched by the highest sediment depths beneath this plant stand; this suggests that a critical storage limit had not been reached. The accumulated bed sediment is an indicator of patch trapping efficiency under varied conditions and over a much longer time-period, and it is encouraging that this coincides quite well with the turbidity results.

6.10.3 Evaluation of the experimental design and suggested improvements

The results from this preliminary study have shown that the experimental design is a viable method for investigating macrophyte interactions at the micro-scale. However, the study has also highlighted a number of limitations in the sampling strategy and several improvements can be suggested to augment the basic design for future research. The most crucial of these changes is the incorporation of a 'mixing length'; a clear, un-vegetated stretch of water of 1 – 2 metres length between the sediment release point and the most upstream turbidity probe. This would ensure

Table 6.8 Measured sediment depth beneath the six selected vegetation stands, based on the average depth as sampled at 0.2 m increments along the central longitudinal axis of the plant stand. Sediment depth was measured before the turbidity experiments were carried out.

Experiment	Average depth (cm)	Standard deviation (cm)
Crockways: small vegetation patch (R1)	4.44	1.87
Crockways: medium vegetation patch (R2)	6.60	3.62
Crockways: large vegetation patch (R3)	3.73	1.41
Hydeclift: small vegetation patch (R1)	6.10	2.76
Hydeclift: medium vegetation patch (R2)	6.67	2.90
Hydeclift: large vegetation patch (R3)	4.09	1.65

that all turbidity readings are taken downstream of the 'mixing distance' and are representative of the sediment event at their location. Most importantly, this would allow a quantified comparison of the incoming sediment load and of the sediment changes occurring between probe locations. However, the inclusion of a mixing length may be difficult to facilitate under field conditions, especially in summer, because *Ranunculus* stands are rarely found in spatial isolation from other stands and are often interlocking in their growth pattern. The removal of neighbouring stands may be necessary to obtain the desired mixing length but this would disturb the natural flow pattern and sediment interactions in the reach and would require permission from both landowners and from the owners of fisheries rights in the river. A second improvement would be to greatly increase the number of turbidity probes in the probe array. This would improve the spatial coverage of the experiments and allow a more precise description of sediment interactions and sediment attenuation within the plants. In particular, a supplementary probe is needed just within the downstream tail of the vegetation to enable separation of sediment attenuation and recombination processes (see Section 6.9.1).

Another desirable change would be an extension of the experiment to cover different periods in the vegetation growth cycle and to include low, moderate and high discharge conditions. This would greatly add to the value and applicability of the results, but would undoubtedly involve considerable field effort. The turbidity experiments may not be possible under all conditions; for example, higher discharges are usually accompanied by high background turbidity and this would make it very difficult to discern the artificial sediment events above the high background levels. Higher discharge experiments may be possible during the falling limb of the storm

hydrograph when turbidity is slightly lower compared to the rising limb. As far as is possible under field conditions, it would also be desirable to have comparable initial velocity conditions for each vegetation stand. Facilitating this may require removal of upstream vegetation stands. A more comprehensive suite of vegetation descriptors is also recommended and should incorporate measures such as vegetation length, height, width, and some measure of vegetation density, such as biomass per area or the leaf area index. Both these density measures require the removal of the vegetation after the experiments have taken place and would not allow repeat measurement of the same plant stand on successive field visits. The vegetation measurements may also be required at shorter intervals; relating to the positions of the turbidity probes for example, rather than as a general summary measure for the whole plant.

Each suggested improvement would necessarily increase the time and effort required for each experiment and an alternative approach might be to test the experimental design in a laboratory. Tests in a flume would allow precise control of all variables and would give much clearer and more definitive results. This would allow regulation of mixing length, vegetation characteristics; water velocity and water depth and would allow greater control of sediment releases. The manipulation of parameters in the laboratory, with stationary equipment, would give substantial time-savings and eliminate the need for a massive field campaign. In addition, the sediment released in the flume, and not retained by the plants, could be filtered from the flow to provide an independent check on the turbidity results. The main difficulty with the laboratory experiments, however, would be the perennial problem

of accurately replicating the complex natural river situation, as revealed in Chapters 4 and 5.

7. CONCLUSION: SYNTHESIS OF RESULTS AND FINDINGS

This final discussion focuses on providing a synthesis of the main findings contained in the four analytical chapters of the thesis. It concentrates on the original thesis aims, as outlined in Section 1.3, and attempts to draw together findings obtained through all measurement techniques and scales of investigation, to present a coherent and comprehensive understanding of the thesis' contribution to addressing the research questions listed in Table 1.1.

7.1 The catchment in context

To place chalk rivers, and more specifically the River Frome and the chosen study sites, in a national context in terms of physical river characteristics and macrophytic vegetation type and abundance.

As listed in Table 1.1, the first aim of the thesis was to define a 'typical chalk river' and to determine how chalk rivers compare to rivers from different geological and geographical areas. Linked to this, this aim set out to discover how the River Frome compared to other chalk rivers and how the chosen field sites compared to other sites on the River Frome.

The RHS analysis provided a quantitative exploration of the full geographical and climatic range of English chalk rivers and put forward the idea that chalk rivers form a distinct UK-river sub-group which may be reliably defined by shared in-channel features. Chalk rivers were defined as having a lower source height, slope and

altitude than rivers of other geology and generally possess a lower energy regime. Chalk rivers also displayed a slightly higher width depth ratio, coarser substrate and generally lower levels of riparian shading than rivers of other geology. These physical traits, and the high nutrient availability in chalk streams, mean that chalk rivers have a higher in-channel vegetation (ICV) index than other rivers and have a higher number of choked sites (vegetation cover greater than 33 %) that are likely to experience plant-induced reductions in mean water velocity and seasonal increases in river stage.

The RHS analysis was also used to compare the River Frome to the chalk average, and this suggested that the Frome is not a typical example of a chalk river. The analysis demonstrated that the Frome has a higher source height and higher slope than most other chalk rivers and therefore has a higher energy regime. Perhaps as a consequence of this higher energy regime, and because the Chalk sites on the Frome have a higher riparian tree cover than other chalk rivers, the Frome has a slightly lower vegetation cover than the chalk average. The Frome also has a wider, shallower, channel shape than the average for other chalk rivers. However, the Frome is more similar in its physical characteristics and vegetation cover to chalk rivers than to non-chalk rivers, and may be considered a member of the UK chalk river group. Research undertaken on the River Frome may be applied to other chalk rivers with the acknowledgment that some characteristics differ from the chalk average, particularly energy regime and channel shape.

The study sites on the River Frome chosen for this thesis were examined using primary RHS survey data to determine if the sites were representative of the River

Frome. Comparisons were made between the three sites and to the Frome average. Given their shared source height, the study sites were generally close to the Frome average in terms of slope, source height and energy regime, but differed from the River Frome average, and to each other, in terms of width/depth ratio, total tree score, in-channel vegetation index and bed substrate. The total tree score and In-Channel Vegetation index are particularly important in the context of this thesis and both differed markedly between sites. The Crockways and Frampton sites were very close to the Frome average in terms of tree score, whereas the Hydeclift Plantation site had very strong riparian shading compared to the Frome average, which inhibited in-channel vegetation growth at the site. The ICV index differed greatly between study sites, the summer ICV index values for the Crockways and Hydeclift sites were below average for the river Frome, while the ICV index for the Frampton site was well above average. However, the in-channel vegetation index did not vary between sites as would have been expected from the total tree score values: the summer ICV index at the heavily shaded site at Hydeclift was higher than at the moderately shaded Crockways site. This suggests that tree cover is not the only major control on in-channel vegetation cover. These defined differences between study sites provided interesting contrasts in vegetation influence and help identify the physical controls that determine the vegetation abundance and influence.

As chalk rivers were shown to share many of the same physical and ecological characteristics, it is likely that they share similar management challenges and should require similar management solutions. The analyses in all four analytical chapters have underlined the importance of providing contextual information alongside measures of vegetation influence. This thesis provides physical descriptions and

contextual information for the collected data and facilitates the comparison of the research findings to different river sites. Contextual information is essential if the results are to be utilised by river managers.

7.2 The stage/discharge relationship

To examine the influence of macrophytic plants on the stage/discharge relationship.

The second thesis aim investigated whether the stage/discharge relationship at the study sites changed in accordance with seasonal changes in plant cover, and, if so, how: does stage increase linearly with increasing plant cover? Or must some critical plant cover be reached before effects are evident? Does stage increase linearly with increasing discharge? Or does the compression of plant morphology and biomass loss cause a change in the stage/discharge relationship at high flow? Do seasonal sedimentation patterns have any effect on river stage? The second aim also investigated whether macrophyte cover can be sufficient to cause a significant increase in the frequency and magnitude of overbank flooding in lowland groundwater fed rivers. Finally, this second aim addressed how the vegetation effects changed from site to site due to differences in morphology, riparian shading and plant biomass.

The PT and grid reach analyses both suggest that vegetation growth can increase river stage and water depth in summer compared to winter at equivalent discharges, but these increases were small (maximum observed increase 0.17 cm) and, due to the stable summer flows experienced in chalk rivers, were very unlikely to increase

either the frequency or magnitude of *overbank* flooding. The observed stage increases was primarily due to the increased hydraulic resistance caused by the vegetation and by consequent decreases in mean reach velocity, but sedimentation changes induce by the plants are also believed to play a smaller part in river stage changes at vegetated sites. A mean *reduction* in bed level was observed at Crockways between winter and summer while a mean *increase* in mean bed level was observed at Frampton. These seasonal bed level changes mean that the influence of vegetation on water depth was underestimated at Crockways and overestimated at Frampton (see Figure 4.27, Chapter 4). Increases in stage and water depth were strongly controlled by physical site factors and the vegetation influence was shown to increase stage at some sites but not at others.

The field sites used in this thesis were chosen to showcase the effects of physical site factors, including contrasts in river planform, water depth and riparian shading. For example, obvious vegetation-induced effects on water depth were evident at the straightened and uniform PT reach at Frampton but were not as evident at the morphologically diverse PT reach at Crockways. Similarly, in the grid-based analysis, obvious water depth effects were evident at the straight, relatively homogeneous and predominantly un-shaded Crockways grid reach but not evident at the straight and extremely homogeneous yet heavily shaded Hydeclift grid reach. These site comparisons demonstrate the importance of channel morphology in controlling vegetation abundance and influence, but also suggest that riparian shadings imparts a similar level of control at less complex sites. A comparison of averaged grid and PT findings also suggests that differences in water depth increases

could be highly localised as changes in vegetation influence were observed over a short distance of ~ 3 m.

The complex morphology at the Crockways PT reach and the strong riparian shading at the Hydeclift grid-reach served to restrict vegetation growth in some areas of the channel and prevent the vegetation from achieving uniform growth. However, some areas of both sites were heavily vegetated and this suggests that the increase in hydraulic roughness and reduction in velocity experienced inside vegetation beds was not sufficient to reduce mean velocity and it is likely that, at the reach scale, increases in velocity outside plant beds were able to compensate for reductions within. The results also suggest that velocity reductions within vegetation stands at the Frampton PT reach and Crockways grid reach were not wholly offset by increases in un-vegetated areas. These site differences were explained through the existence of a 'critical biomass': a specific vegetation cover after which compensatory increases in velocity outside of plant beds are no longer sufficient to offset the velocity reductions within plant beds. Once this critical biomass is achieved then mean reach velocity is reduced and river stage is elevated. This critical threshold vegetation cover is likely to have been achieved in late spring at both sites when a rapid switch between the winter and summer stage/discharge regimes was observed.

The agglomerative grid analysis further revealed that increases in water depth were not constant following the attainment of the first critical biomass as was assumed from the seasonally averaged PT trends. A critical biomass is needed for the initiation of vegetation effects but vegetation influence and water depths continue to

increase throughout the growing season as vegetation becomes more abundant. The cluster analysis suggested that the increase in river stage may have continued in a gradual and incremental manner until late summer when a second critical biomass was achieved. This second threshold is thought to represent the closing of preferential flow channels by encroaching vegetation. The blocking of compensatory preferential flow channels means that all areas of the channel experience decreased velocities relative to the winter baseline and this leads to a dramatic reduction in average reach-scale velocity and a large increase in water depth.

Vegetation induced increases in water depth were also shown to be non-linear and to vary with discharge. In the PT analysis, water depth was observed to increase at a higher rate with increasing discharge in summer than in winter (using both net and proportional stage comparisons) and this implies that stage increases are highest at high discharges. This positive trend in stage is mirrored by the relationship between water depth and discharge and further validated by trends in velocity. These trends in river stage, water depth and velocity do not comply with the theories of Watson (1987) that a reduction in hydraulic roughness, an increase in water velocity and a decrease in river stage will occur at higher discharges. The thesis findings suggest that vegetation effects on stage may be increased at high flow, perhaps because of increased contact and interaction of *emergent* vegetation with water flow at high discharges. Discharges in the Frome may not be sufficient to cause a constriction of submerged plants, or biomass loss, or perhaps the increased roughness generated by the submergence of *emergent* plants is greater than the decrease in roughness caused by the compression of submerged vegetation. Watson suggested that the n-VR curve

was only strictly applicable to submergent plants and the results found in the current research reflect the combination of submergent and emergent species effects.

The site differences and non-linearity revealed in the present study, and the wide ranging estimates in the published results, highlight the complicated nature of vegetation influence on river stage and other reach-scale hydraulic parameters. The analysis has revealed that the stage increases at a site will be determined by the value of the critical vegetation biomass and whether or not this critical biomass is attained. If the critical biomass is attained in a reach, then the strength of the stage increase will be controlled by several different physical factors: the vegetation abundance at a site, which itself is controlled by channel morphology and riparian shading; vegetation type; channel shape (the width/depth ratio); seasonal changes in sedimentation and the range of discharges over which measurements are taken.

7.3 Flow resistance and velocity

To investigate the effects of macrophytes on microscale and reach-scale flow resistance and velocity.

The third thesis aim addressed the extent to which velocity was reduced inside macrophyte beds and whether a compensatory increase in velocity could be observed outside plant beds. The main purpose of this investigation was to determine whether increases in velocities outside of plant beds were sufficient to compensate for decreases inside plant velocities and whether overall reach velocity would be affected by vegetation influence. A central issue to this was whether the

compensatory action outside plant beds is subject to a maximum biomass limit, after which increases in velocity outside of plant beds can no longer compensate for decreases within. Related to this, the question was posed whether self-limiting of biomass by macrophytes, riparian shading or complex channel morphology could help keep aquatic biomass below this critical level.

The seasonal trends explored in the PT analysis showed that mean velocity was greatly reduced in summer at Frampton but the difference only slightly reduced at Crockways. The agglomerative grid data revealed similar site differences: summer reductions in re-scaled velocity were stronger at the Crockways grid site and less marked at the Hydeclift grid site. These site differences can be explained by differences in channel morphology and in riparian shading which control macrophyte abundance and are capable of maintaining vegetation biomass below the critical thresholds. The self-limiting of biomass was not able to keep macrophyte cover below the critical threshold and, indeed, macrophyte biomass, velocity reduction and sage increases continued throughout the growing period at the Crockways grid reach, achieving a first and second critical biomass and only ultimately limited by climatic changes and the senescence and washout of the plants.

The grid-based analyses in Chapter 5 explored the small-scale interactions between vegetated and un-vegetated areas which determined the extent to which mean velocity was reduced and stage was increased. Hierarchical cluster analysis of the grid data explored the relative influence of vegetated and un-vegetated clusters on mean reach velocity and suggested that the characteristics of cluster cells are determined by the vegetation cover within their cells and by the vegetation cover in

the reach as a whole: high vegetation cover in one section of the channel will reduce velocity in this area and will increase velocity in compensatory clusters of low vegetation cover. The cluster analysis revealed that there are two main factors which determine whether critical biomass is attained and whether stage increases are realised in a reach. These factors were: (i) the size of each cluster; and (ii) the degree of seasonal change in each cluster. Cluster size, i.e. the area of the channel covered, is likely to be the dominant factor in any reach, but the dominance of the largest cluster may be accentuated or undermined by differences in the degree of seasonal velocity change between clusters.

At Hydeclift there was a simple distinction between the left and right sides of the channel, imposed by riparian shading. The un-shaded and vegetated right side of the channel experienced marked reductions in water velocity, but this was adequately compensated for by increases in velocity in the shaded and un-vegetated left side of the channel. Though the reduction in velocity in the vegetated cells was greater than the increase in velocity observed in the un-vegetated cells, the area of the shaded section was greater and the dominant influence of cluster size meant that overall mean velocity was not reduced and river stage did not increase. At Crockways a more complex system of ribbon-like preferential flow-channels was revealed. The flow channels occurred in transitional areas between emergent and submergent vegetation, where conditions are least favourable for colonisation. These flow channels are initially able to compensate for the decrease in velocity observed within plant beds, but, as biomass increases, and vegetated cells begin to outnumber un-vegetated cells, increases in water velocity in the preferential flow channels is no longer sufficient to override the effects in the vegetated cells. Mean velocity is

reduced and river stage increases. This marks the first critical biomass, in late spring.

The cluster analysis also revealed a possible second critical biomass in August at Crockways, which led to a dramatic reduction in average reach-scale velocity and a marked increase in water depth. At the second critical biomass each cluster, vegetated and un-vegetated, exhibit a reduction in velocity and it is thought that the preferential flow channels become blocked by encroaching submerged vegetation. There is evidence that the deeper 'pool' cells in the Crockways reach show an increase in velocity after the second critical biomass as the flow channels become blocked. Under un-vegetated conditions pools act as a diversionary feature: most water approaching the pool accelerates around areas of deep, slow water rather than travelling through them. However, when the faster-flowing preferential flow channels are closed, the low vegetation cover in the pool means that it becomes 'activated' as a new preferential flow route. This suggests that there is a 'depth limitation' to vegetation effects before the second critical biomass is attained. This activation of deep water cells was not sufficient to prevent dramatic increases in river stage in the Crockways grid reach, but is likely to be an important control on stage increases in a reach with a larger pool feature or several small pools.

The grid-based analyses also revealed that velocity variability can be reduced in summer at high biomass sites, which is in opposition to the findings of previous research (Jones *et al.*, 1994) that suggests that vegetation creates more varied flow conditions and habitats suitable for a greater diversity of biota. The findings from this thesis suggest that the establishment of submerged *Ranunculus* plants in fast

flowing areas acts to slow the flow in these areas and, after a critical biomass is reached, acts to accelerate the flow in deep slow flowing areas. At Hydeclift velocity variability is greatest in summer, as the first and second critical biomass thresholds were not met. It would seem that velocity heterogeneity and habitat diversity increase in vegetated streams until the second critical vegetation biomass is attained and after this point velocity variability is progressively reduced and may be reduced beyond that which exists under winter conditions.

7.4 The shifting distribution of fine sediments and factors controlling sediment retention in macrophyte beds

To summarise the shifting distribution of fine sediments, both seasonally and spatially, at the meso and microscale. To assess the impacts of macrophytic vegetation on sediment processes and identify the mechanisms that control sediment retention within macrophyte beds.

The fourth thesis aim sought to identify the preferential depositional areas for fine sediment in each season and to record the changes in the depth of sediment retained through the year. The critical questions centred on whether sedimentation was increased within macrophyte beds relative to un-vegetated areas and what were the main mechanisms that brought about increased sedimentation within the macrophyte beds, e.g. were erosion rates reduced or were depositional rates increased? Or, did both processes combine to increase sediment retention in macrophyte beds? Sediment processes were investigated at each of the three research scales: using agglomerative reach scale data across the PT reach and grid reaches; segregated data from the grid reaches; and microscale turbidity data. Seasonal changes in sediment

scour and fill across the whole river reach were examined in the PT analyses and, like stage changes, the data revealed significant differences between study sites.

At Frampton, the reach-averaged bed level in summer was 0.04 m higher than the winter level and this suggests that sedimentation within vegetation beds at Frampton *is* capable of increasing overall mean bed level: the increases in bed level within macrophyte beds were higher, or covered a larger area, than reductions in bed level which took place in un-vegetated areas. Personal observations at Frampton and published data from LOCAR (Cotton *et al.* 2006) suggest that fine sediment accumulation within plant beds can reach depths greater than 0.04 m and that erosion in un-vegetated areas of the reach was able to partly offset the increased sedimentation within plant stands at the reach level. In contrast to Frampton there was a small *decrease* in mean bed level of 0.1 m at Crockways between winter and summer. This was thought to be a consequence of high sediment retention loads retained in the high biomass grid reach immediately upstream of the PT cross-section. The high sediment retention rates upstream meant that either sediment supply is reduced in the un-vegetated reaches or that, having deposited much of its sediment upstream, the water has more power to erode the bed in the sparsely vegetated reach. If viewed as a proportion of total seasonal stage change, the estimates of bed level change reduce the water depth increases caused by increased hydraulic roughness at Frampton and increase the water level depth at Crockways.

The agglomerative analysis of grid-data did not reflect the seasonal changes evident in the PT data and instead indicated that sediment retention is heavily influenced by antecedent discharge conditions over short timescales: the variation evident between

individual surveys in each season was greater than the variation between seasons. The differences between the two patterns may be explained by the methods by which the data were obtained. The seasonal sediment changes revealed in the PT data were based on continuous results obtained over a two year period which were able to smooth the effects of high discharges to allow evaluation of seasonal trends. The grid measurements, by contrast, have a higher spatial coverage but rely on only nine non-continuous data sets. The snapshot grid measurements are highly vulnerable to the influence of antecedent discharge conditions. The segregated grid data were primarily designed to investigate smaller scale patterns of seasonal change and to allow comparison between vegetated and un-vegetated areas. However, the grid resolution of 1 m² proved too coarse to be able to differentiate the sedimentation in vegetated and un-vegetated areas and this compromised the utility of the results. There was some indication that sedimentation was higher in vegetated than un-vegetated areas, particularly at the Hydeclift site, but it was unclear whether this was caused by an increase in sediment deposition beneath plant beds or by a concurrent increase in erosion in un-vegetated areas.

The microscale analysis built on knowledge from the PT analysis that sediment was preferentially retained within macrophyte beds and explored the processes which control the deposition of fine sediment within individual macrophyte beds. The microscale experiments tested the suitability of vegetation length as a possible proxy for the expected degree of sediment attenuation but the results show that length is not a single definitive measure of sediment attenuation; instead each individual plant will interact differently with water flow and with sediment depending on: maturity; length, width and density; initial water velocity; water depth; the type of suspended

sediment and the proximity of neighbouring vegetation. The experiments showed that each vegetation patch is a unique filter, with complex and variable characteristics: the characteristics of each filter change over the growing season and vary throughout the length of the plant.

The microscale turbidity experiments revealed that the amount of sediment retained by each plant was determined by the balance between sediment ‘attenuation’ and ‘diversion’, and that the amount of trapped sediment may be *less* in larger plants than smaller plants. The greater velocity reduction experienced in large plant stands creates a stronger barrier to flow and sediment transport than smaller plants and this means that more flow is diverted around the plant and attenuation is less than for smaller plants where more water and sediment is able flows through and into the vegetation. This balance may shift at a critical mass is also likely to differ between plant stands of similar vegetation characteristics (density maturity length, etc.) but with different initial water velocities and may differ at the same vegetation stand if discharge changes.

7.5 The pore size of the vegetative filter

To describe the sediment filtering effect of macrophyte plant beds - to determine whether deposition within macrophyte beds is a size selective process and to define the 'pore size' of the suspected vegetative filter.

The fourth thesis aim sought to determine whether the observed sediment retention within macrophyte beds size was selective, i.e. were some particle sizes retained by

the plant and others not retained, and whether both suspended sediment and bedload contribute to deposition within macrophytes. This aim also sought to determine whether an upper limit of sediment retention exists and whether the length of a macrophyte stand affected the filter characteristics or the filter capacity.

Results from the microscale investigations suggested that the retention of fine sediment within macrophyte beds is size selective. An examination of the sediment stored within the canopy of several macrophytic plants provided quantified size ranges for retained sediment and demonstrated that the size range differs depending on the architecture of the plant, the shape of the plant leaves, the position of the plant in the channel, and antecedent discharge conditions. For fine-leaved submerged plants such as *Myriophyllum alterniflorum*, and analogously *Ranunculus calcareous*, a change in the characteristics of sediments was observed with increasing distance downstream; large particles were preferentially trapped in the central sections of the plant while the highest volume of smaller particles was found the downstream section of the plant. When the plant was considered as one whole sample three overlapping sediment size distributions were observed, at: (i) $\sim 20\ \mu\text{m}$; (ii) $\sim 160\ \mu\text{m}$ and (iii) $\sim 950\ \mu\text{m}$ (described by the mode of their peaks). These three distributions may be a reflection of the particle size distribution of the source material available in the river; a response to differential particle trapping by different parts of the plant; patterns of sedimentation longitudinally through the plant, in response to velocity changes; or may represent vertical zones within the plant from the water surface to the bed. This last explanation may itself reflect the height of suspension of different particles in transport and the frequency of suspension of the different particle sizes.

Evidence of size selective sediment retention was also demonstrated by the turbidity experiments: the larger sediment grades showed less response downstream than the smaller particles and suggests that the larger sediment particles are retained in greater volume than the smaller particles. Similarly, in the analysis of the peak magnitude of the sediment events, the overall attenuation of sediment was greatest for the larger sediment grades and least for the finer sediment grades. All sediment grades showed some attenuation of sediment meaning that a range of particle sizes are retained but in varying proportions.

When considered in tandem, the washing and turbidity experiments suggest that sediment trapping by macrophytes involves both suspended sediment and bedload. The particle size range of the introduced sediments were based on the range of sediments found in the washing experiments but when these were introduced into the flow in the turbidity experiments they produced little downstream response. This suggests one of three explanations: (i) that despite similar plant morphology and architecture there are significant differences in the sediment accumulation within *Myriophyllum* and *Ranunculus*; (ii) that coarser sediments enter the plant canopy predominantly as bed load (Cotton *et al.* 2006; Wharton *et al.* 2006) and are not identified by the turbidity experiments; or (iii) that coarser sediments do enter the plant canopy as suspended sediment but only under high discharges.

7.6 Potential management implications of the research findings

To contribute to the scientific base needed for the sustainable management of lowland groundwater-fed river systems. To provide empirical data to help inform

river managers of the optimum macrophytic vegetation cover in streams and of the appropriate timing and methods of management.

The final thesis aim links all the previous research questions outlined under aims one to five and seeks to align these findings towards possible management recommendations. These suggested improvements and refinements are centred on possible controls on vegetation biomass: when is the most effective time for management intervention and what alternatives to traditional management can be suggested on the basis of the research findings. Central to the recommendations is the consideration of critical biomass: the threshold level of vegetation at which summer stage increases occur.

The thesis research findings may be used to strengthen the scientific base available to river managers and to suggest changes to current management regimes and methods. Overall, the most oft-cited reason for aquatic vegetation control is for flood control and this section concentrates on measures that can help provide a compromise between vegetation management for flood control and for maintaining and maximising biodiversity. In particular, river managers need to implement management schemes that consider sites according to influencing physical factors and the abundance of vegetation. A blanket approach to macrophyte management along a large section of river will be successful at some sites but not at others, and management may not even be necessary at some sites. Two main methods of macrophyte control are considered: (i) temporary and repeated management and (ii) 'permanent' management.

7.6.1 Temporary and repeated management

Current management regimes concentrate on temporary and repeated measures to manage aquatic macrophyte populations, primarily achieved through manual cutting of the vegetation and with an emphasis on flood control. The holistic research findings outlined in this chapter summarise new insights into macrophyte influence that could help improve the timing and methods of this temporary management, making management regimes less ecologically damaging *and* more time and resource efficient.

Traditional management usually involves the complete cutting and removal of vegetation along large sections of a river where there is a perceived flood risk (Holmes, 1999). The concept of a critical biomass espoused in this thesis suggests that not all sites will require management. Physical factors, e.g. riparian shading and in-channel morphology, may be such that the vegetation biomass remains naturally below the critical threshold. These site differences suggest that a blanket management approach applied to large sections of the river will work at some sites, may not work at others, and will not even be necessary at some sites. This means that resources may be wasted by cutting vegetation where no cutting is needed. The research findings provide some guidelines as to which sites may require and may not require management, e.g. cluster analysis of the grid data suggested that critical biomass is likely to occur if more than half the channel at a site is heavily vegetated and analysis from both Chapter 4 and 5 suggest that physical factors can control vegetation biomass. The estimates of stage increase in this thesis are given in their physical context and this allows the estimation of stage impacts on other rivers, e.g.

if the width/depth ratio is lower at a proposed management site than the width/depth ratio at Frampton or at Crockways then stage increases are likely to be higher at this site (if all other factors are equal).

Sites which may require management could initially be differentiated by desk studies (possibly using the RHS database), which examine the morphology of the reach and the degree of riparian shading, and could be refined through field surveys of vegetation in selected reaches. If possible, the installation of a simple discharge cross-section station, fitted with a stage board would allow comparison of stage height and discharge and provide definitive evidence of the nature of the vegetation impact on stage at that site. The PT analysis showed that increases in stage were consistent over two water years and this suggests that the initial viability study need only be completed once, with periodic reviews of river and site changes.

For sites with genuine flood control needs, the cutting regime at these sites may also be refined using findings from the thesis. Vegetation in a reach could conceivably be cut to below the critical biomass of a site, cut in some areas and not in others, and still achieve the same reduction in flood risk. However, achieving a sub-critical biomass across the channel would be difficult and expensive to maintain (van Nes *et al.*, 1999 and 2002) and it may be more feasible to implement zoned vegetation cutting: allowing some areas to be completely cut while others are left untouched. For example, the research findings suggest that vegetation effects on water depth may be highly localised and high vegetation biomass in one channel cross-section may affect water depth in this cross-section but not affect water depth a few metres downstream or upstream. If vegetation effects on water depth are only able to build

up across a series of vegetated cross-sections, then management regimes could be established that alternate even quite short sections of cut and un-cut vegetation to obtain the same reduction in flood risk as would a full cut. However, the exact extent to which vegetation effect on water depth propagate upstream and downstream could not be determined by the methods of this thesis and further research is required. It is thought that vegetation effects may propagate further upstream than downstream and this may complicate the implementation of this management method.

The pattern of cutting in a river reach may be especially important. The research findings suggest that preferential flow channels of un-vegetated space are very important in moderating vegetation effects and these may be created by cutting vegetation to produce an un-vegetated ribbon of channel. This would be similar to the grid reach at Hydeclift where high biomass occurred in un-shaded areas at the right bank but the effects of this were 'cancelled out' by un-vegetated areas at the left bank. To avoid preferential cutting of any one vegetation type or species, it might be preferable for the un-vegetated ribbon to alternate from bank to bank in a meandering fashion and not merely cut as a swathe through the thalweg of the channel. This arrangement of cut and uncut areas would negate the problem of uncertain distances of vegetation influence between cut and uncut areas upstream and downstream.

The timing of the vegetation cut may also be very important. The optimum timing of a vegetation cut depends on the degree of summer water depth increases that would be acceptable to the river manager. If no water increase is permitted, then vegetation must be cut before the first critical biomass is attained at the site. The research

findings indicate that the critical biomass occurs during April at the Frampton PT reach and between April and May at the Crockways grid reach. This timing will vary slightly from site to site and river to river but is likely to occur in late spring at most English chalk river sites. The timing of this cut is earlier than the usual timing of vegetation cuts reported by most authors (Westlake, 1968; Robson, 1974; and Soulsby, 1974) and may mean that a second cut is required later in the year when vegetation biomass recovers. If small increases in water depth can be tolerated by the river manager then it may be more logical to cut vegetation just before the theorised *second* critical biomass. This second biomass marks the point when preferential flow channels are closed and water depth levels increase markedly. This second biomass was observed at the Crockways grid site in August. This more tolerant approach would accommodate a much later cut and is likely to only require one cut per year.

7.6.2 'Permanent' management

The research findings also point to the possible viability of a new management approach: a switch from traditional, temporary and routine management to 'permanent' management solutions. These include: (i) the planting of riparian trees to increase shading of the channel and restrict plant growth; and (ii) the wholesale restoration of channelised river reaches to provide more morphological diversity and diversity in vegetation type and abundance.

The comparison of shaded and un-shaded reaches in Chapter 5 suggested that riparian shading of the channel can keep vegetation abundance below critical

biomass. These findings support the work of Dawson (1978) and Dawson and Kern-Hansen (1979) who suggested that riparian shading can be used as a management tool. The planting of riparian trees at problem sites would recreate the natural situation at Crockways, but unless mature trees are used then it would take time for trees to become established and for the shading benefits to be realised. Temporary cutting measures may need to continue at the sites before the tree cover becomes established. Dawson (1978) set out criteria for the appropriate shading at various river sites which differed depending on the orientation (e.g. north-south or east-west) of the river reach and on which bank riparian trees occur, i.e. trees on the south bank would presumably cast more shade than trees on the north bank, while west and east banks would be approximately equal in their effect. The current research findings add to these criteria: the Hydeclift reach is shaded only on the left bank and this suggests that tree cover does not need to be total and together with the cluster analysis findings suggests that only 50 % shading of the channel is required. It might also be feasible to create a zoning of shaded and un-shaded reaches, similar to the cutting method described in Section 7.6.1: stretches of riparian tree cover could be interspersed with open un-shaded river sections. Tree planting on the river bank may have several attendant benefits in terms of bank protection, habitat creation and aesthetic value.

Another possible and more radical management tool suggested by the research findings is the broad-scale rehabilitation of artificially straightened, homogeneous and highly vegetated channels. The difference in vegetation influence between Crockways and Frampton was thought to be dictated by the channel morphology: Crockways represents a semi-natural river channel with minimal human intervention,

whereas Frampton represents a channelised reach from which natural planform and in-channel features have been removed. If the high biomass at Frampton is viewed as a consequence of its unnatural state, then river restoration and the reinstatement of in-channel features may serve to reduce biomass through depth and velocity changes and reduce, or even negate, the influence of vegetation on water depth. River restoration to control vegetation would provide enhanced habitat and aesthetic value and would provide an economic incentive for restoration.

7.7 Suggestions for further research

The thesis has examined many aspects of vegetation influence in rivers and has, to a greater or lesser degree, addressed many of the uncertainties set out in the original thesis questions. The analyses have emphasised the importance of repeated measurement over long timescales, including winter baseline conditions, and the importance of conducting measurements at several and temporal spatial scales so that one scale may inform another to give a more holistic picture of the research problem. However, the research has also revealed the complexity of macrophyte influence. Further questions have arisen which require further research.

7.7.1 Extension of spatial and temporal scales: future detailed research

Several hypotheses have been put forward that require further substantiation and would generally require an extension of the research on both temporal and spatial scales. For example:

- i. An increase in the temporal periodicity of discharge measurements would enable more precise measurement of the transition period between summer and winter stage/discharge regimes and give an indication as to the rapidity of the transition in spring and in autumn. Another useful extension would be the repetition of measurements in a different water year to investigate the constancy of the timing in each study reach and the factors that might affect this.
- ii. An increase in the spatial distribution of the PT network would help determine how connected or localised the vegetation signal is. For example, over what distance does the influence of a vegetated section of the channel propagate a) upstream and b) downstream in the river reach. Use of stage and not water depth as used in the grid analysis in Chapter 5 would negate the problem of possible bed level change between surveys.
- iii. The seasonal patterns of sediment storage differ greatly between the PT and grid methodologies. An increase in the number of grid-surveys and a decrease in the time period between surveys are recommended for further research to reduce the influence of antecedent discharge conditions. In addition, several methodological problems were outlined within the sediment analysis in Chapter 5 and the sediment measurements could perhaps be repeated after implementation of these changes.
- iv. The cluster analysis in Chapter 6 concentrated primarily on physical flow parameters and compared the observed patterns in velocity data to the spatial and temporal patterning of the raw vegetation cover data. It would be possible to

include both velocity data and vegetation cover data within the same clustering procedure to obtain a more integrated analysis of seasonal macrophyte-velocity interactions. However, the data obtained from such a clustering procedure would be highly complex and would require careful interpretation.

- v. Further research is required at the microscale using the high frequency turbidity experiments piloted in Chapter 6. In particular, a more sophisticated experimental design is required to assess the repeatability and transferability of results: it is possible that the complex variations revealed in the experimental data may belie more consistent trends or patterns at larger-scales if more information were available. Suggested improvements for the experimental design have been outlined in Section 6.10.3 in Chapter 6.
- vi. The washing experiments undertaken in Chapter 6 were included to illustrate an experimental design that was used to inform more detailed microscale research. This sampling method should be repeated with the detailed assessment of the sediment components as the primary objective of the investigations. This would mean taking account of the volume of both the organic and inorganic components of the samples and measuring the “effective particle size” of the samples, i.e. including fine particulate organic matter and aggregates of organic and inorganic particles, following the methods of Wharton *et al.* (2006). The sampling should be extended to several sites, at different times of the year.
- vii. Application of the methods from the thesis to other chalk rivers and to rivers of other geology is recommended. For example, rivers with flashier, higher

discharge regime would be useful in examining the validity of Watson's n-VR model. The model does not appear to hold true in the Frome but it may be valid at higher discharges than those encountered in the Frome.

One general suggestion for future research would be to stress the importance of providing contextual information about the physical characteristics of a study site. Site differences on the River Frome were such that seasonal water depth changes arising from vegetation influence of were entirely inconsistent unless physical factors were taken into account and the comparison of the current research findings to those from previous research was undermined by the lack of contextual information available in previous studies.

7.7.2 Streamlined research methods: viable methods for routine management

Many of the data collection methods deployed in this project are not suitable for replication as part of routine management. The methods generally require much time in the field and employ expensive equipment. However, the results obtained through these data collection methods can help suggest whether more simplified methods would give accurate results and may point to more streamlined methods of data collection and exploration.

The River Habitat Survey database analysis in Chapter 3 proved capable of characterising river sites. In particular, the database may be used to investigate factors which may be used to measure and predict vegetation abundance at a site: the In-Channel Vegetation index as an estimate of vegetation cover in a reach; total tree

score as a measure of riparian shading and the width/depth ratio as an assessment of channel shape. The RHS database could also be used to characterise the morphological diversity of a reach; for example the RHS survey contains measures of channel features present at spot checks, the number of riffles and pools in a study reach and whether the channel has been re-sectioned. These four RHS derived factors (vegetation cover, riparian shading, channel shape and morphological diversity) could be used as part of the suggested desk study suggested in Section 7.6.1 as a preliminary step in planning macrophyte management. RHS data could help suggest whether vegetation biomass at a site is likely to reach the critical vegetation biomass required for plant-induced water depth increases. The RHS also contains information as to the land use at river sites and this may be used in the desk study to determine the likely economic implications of water depth increases in summer.

The cross-sectional stage/discharge measurements employed in Chapter 4 have proved very insightful and required only modest field effort post installation. Budget considerations aside, this method would lend itself to widespread and simultaneous deployment at several sites. This method may be used to estimate the seasonal changes in water depth at a site and to determine whether management is needed. Several adjustments could make the method more suitable for use in management data collection. The expensive and specialised PT equipment could be replaced with a simple graduated stage board from which manual measurements of the water level could be made during each discharge measurement. The ADV used to make discharge measurements is a very expensive and sophisticated device and less expensive equipment may be used (for example an electromagnetic current meter)

which can be used in the field with minimum training. Research findings from Chapter 5 suggested that seasonal changes in water depth may vary greatly over very short distances and it is recommended that several cross-sections be chosen at regular intervals throughout a study reach. The measurement of stage discharge at closely spaced cross-sections would require only one discharge measurement for the reach, and though it would require a stage measurement for each cross-section this would not be a time consuming procedure. It is suggested that as many stage/discharge measurements as possible be completed over summer and winter but it is likely that less measurements be required than those captured during this research.

7.8 Integration with LOCAR

As outlined in Chapter 1, the work in this thesis formed part of the wider Lowland Catchment Research programme, which facilitated the interdisciplinary study of three permeable catchments. LOCAR projects were extremely diverse and cover the transport and transformation of water, chemicals and sediment across the catchment, both above and below ground, and through river systems to provide an integrated picture of catchment processes.

At the catchment scale projects examined evapotranspiration across different vegetation types, the movement of groundwater and chemicals through the chalk rock and traced the source of sediments within each catchment and developed a sediment budget for each river (LOCAR, 2006). The groundwater chemicals and sediment come together in the river channel, particularly at the river bed and several projects investigated the physical and chemical properties at the interface between

the river channel and groundwater flows: in the riparian zone and at the channel bed. This research suggested that the most important area for exchange between the river channel and groundwater occurred at the surface of the channel bed, within a thin active layer 10 – 20 cm deep (LOCAR, 2006). This layer is heavily influenced by in-stream vegetation and the LOCAR findings from this and other projects (Cotton *et al.* 2006; Gurnell *et al.* 2006; Wotton *et al.* 2006) suggest that plants are river engineers which are instrumental in determining river health. LOCAR has helped disentangle the complex interactions which occur between water velocity, sediment and nutrients.

Other LOCAR researchers have helped to confirm the findings in the present study and have contributed significant new information regarding the role of macrophytic vegetation in controlling bank erosion through deflection of water flows (Gurnell *et al.* 2006), the amount of sediment accumulated beneath macrophytic plants (Cotton *et al.* 2006; Wharton *et al.* 2006), the particle size distribution of accumulated sediment (Cotton *et al.* 2006; Wharton *et al.* 2006), the role of suspension feeders in creating aggregate sediment particles and the need to measure effective particle size (Wharton *et al.* 2006), the steady movement of sediment through plants under base flow conditions and the flushing of sediments under high flows (Cotton *et al.* 2006) and the seasonal differences in seed and propagule transport (LOCAR, 2006). Together these studies have contributed greatly to the understanding of the seasonal effects of large aquatic plants on water velocity, fine sediment and river stage.

Synthesis of the results from these different projects, and from the wider LOCAR investigations has to be achieved across varied river sites and catchments. A

framework for integration is required which must include contextual information which can link and compare sites. The river reconnaissance (Chapter 2) and River Habitat Survey analysis (Chapter 3) conducted in this thesis provides a quantitative context for this project and for other LOCAR projects on the River Frome, particularly those concerned with macrophyte, sediment and velocity interactions (Cotton *et al.* 2006; Gurnell *et al.* 2006; Wharton *et al.* 2006), allowing results from these intensive studies, of limited spatial scope, to be extended to other sites and other catchments. The methods used here could be extended to the other rivers investigated under LOCAR to provide a contextual link between sites within these catchments and to integrate the research undertaken across the three LOCAR catchments.

LOCAR was developed in partnership with regulatory agencies and the overriding aim of programme was that findings of the research provide a scientific underpinning upon which to formulate and implement appropriate and sustainable management policies and practices. The methods outlined in 7.7.2 of this chapter provide practical methods for monitoring vegetated sites and the recommendations in Section 7.6 have relevance to the implementation and evaluation of the EU Habitats Directive (92/43/EEC), the EU Water (2000/60/EC) Framework Directive and local, regional and UK Biodiversity Action Plans (BAP) and, in particular, the chalk river habitat action plan.

APPENDIX A: A BRIEF REVIEW OF MACROPHYTE ECOLOGY

A brief introduction to macrophyte ecology is provided to supply background information essential to the understanding of later chapters in the thesis. For example, it is important that the seasonal growth cycle is understood because the winter channel conditions will be used as a baseline to judge summer vegetation effects. Site differences are also extremely important in this study and an understanding is required as to why plant cover varies from site to site and why some plant species will thrive at some sites and yet be absent at others.

Factors determining vegetation abundance and location in the channel

As noted in Table 1.3, different growth forms tend to occupy characteristic locations in the river channel and previous research has suggested that macrophyte species respond to environmental gradients, for example: light; water chemistry; flow velocity; substratum composition; competition; and biotic factors (Westlake, 1975). In most cases the presence and abundance of a certain species at a site is determined by a number of factors which are difficult to separate, especially as some factors act in combination, e.g. water velocity may also determine substrate type. Haslam describes a combination of physical, chemical and biotic factors, "any of which may be of overriding importance" (Haslam, 1971), though physical factors are thought to be of greater importance and velocity is often thought to be the prime regulator of vegetation growth (Chambers *et al.*, 1991). However, different plants respond differently to each influencing factor.

The two dominant species in the study reaches, *Ranunculus calcareous* and *Sparganium erectum*, differ in their range within the UK and generally occupy very different positions in the channel cross-section, as a response to environmental controls. *Ranunculus calcareous* is a species that is almost exclusively found in calcium rich waters draining either chalk or carboniferous limestone and favours alkaline water of pH 7 – 9 and 100 – 300 mg l⁻¹ of Calcium carbonate (Rodwell, 1995). The position of *Ranunculus calcareous* within the channel is primarily determined by water depth, water velocity and bed substrate, this species is most common in deeper, fast flowing water and coarse substrate in the centre of the channel. Previous research has suggested that the biomass of *Ranunculus calcareous* is linearly related to water depth, with biomass declining from an average of 250 g m⁻² dry weight in shallow water less than 0.5 m deep to 25g m⁻² dry weight in deep water 2.75 m deep (Ladle and Casey, 1971). In very shallow channels *Ranunculus* may have to compete with emergent species such as *Rorripa*. In common with most aquatic plants *Ranunculus calcareous* biomass may also be greatly affected by light availability (Dawson, 1976 and Dawson and Kern-Hansen, 1979).

Sparganium erectum is not restricted in terms of catchment geology, and occurs across a wide area of the UK, but is a common plant in chalk rivers. It favours shallow water and grows best in water 10 – 20 cm deep with silt substrate but may be found in depths up to 1 m (Preston and Croft, 2001 and CEH, 2004). The plant has only shallow roots and cannot grow well in fast flowing water and while it can tolerate some emersion it cannot do so for prolonged periods. This physical niche

means that it is generally confined to the margins of the river. *Sparganium erectum* affected by light availability and shading of the channel.

In river environments a large-scale longitudinal environmental gradient also exists, may occasionally populate the centre of shallow river channels if water flow is impeded by other plants such as *Ranunculus calcareous*. *Sparganium erectum* is also formed from upstream to downstream, such that, in a generalised river, species composition changes with distance downstream, generally from emergent-dominated to submergent-dominated. Biodiversity also generally increases with distance downstream (Haslam and Wolsley, 1981). This downstream change is explored for chalk rivers in Chapter 2 of the thesis.

Macrophyte growth and reproduction

Most macrophytes are herbaceous perennials and each plant will live for several years. In warm, constant climates perennials can grow continuously, but in temperate climates growth is restricted to the warmer 'growing season'. Deciduous perennials experience a period of luxuriant growth in spring and summer, and die back to their roots in late summer and autumn. Dead plant material is washed from the channel during autumn and winter high discharges and the growth cycle begins again with re-growth from existing roots and stem tissue. Differences in species' responses to environmental gradients of temperature, light availability and discharge lead to differences in life cycles between plants and peak biomass may be achieved later or earlier in the year depending on the macrophyte species. Macrophytes may reproduce both sexually and asexually. The method open to each species generally

depends on its growth form, for example many submerged species do not possess aerial reproductive parts. Sexual reproduction occurs as in terrestrial plants, with aerial cross pollination by flying insects and wind pollination, whereas asexual, or vegetative, reproduction occurs when parts of existing plants are transported to new sites in the river.

The growth cycle and annual production of *Ranunculus calcareous* has been studied in great detail in previous research (Ladle and Casey, 1971; Dawson, 1976; Dawson and Kern-Hanson, 1979; Dawson, 1980). Four distinct periods of the *Ranunculus* growth cycle have been identified: (i) re-growth in autumn; (ii) extension phase in late winter; (iii) consolidation and flowering in spring; and (iv) decline in late summer (Dawson, 1976). Re-growth begins after silt and dead plant material are washed out of the reach by autumn and winter high discharges but biomass does not begin to increase rapidly until late winter and early spring. Maximum biomass is generally achieved in June, approximately a month after the plant flowers. The date of flowering is important because the morphological changes required to produce aerial flowers makes the plant stems more brittle and encourages the loss of plant material. However, the date of flowering is thought to vary from the source to the river mouth by 2 – 3 months (beginning first closest to the source) and this means that the timing and extent of maximum biomass may also vary from site to site down the river (Dawson, 1980). After maximum biomass, three-quarters of plant material is lost by the end of September (Ladle and Casey, 1971) and the remainder is lost in autumn flood flows. *Ranunculus calcareous* flowers are produced on buoyant pedicles above the water surface and are fertilised by flying insects, but seed production is low (Dawson, 1980) and the species relies heavily on vegetative

reproduction to produce new plants in the reach and further downstream. Due to its central location in fast currents *Ranunculus calcareous* is susceptible to mechanical breakage and healthy plant propagules may be transported downstream to colonise new areas.

Sparganium erectum exhibits a similar growth cycle. The leaves of *Sparganium erectum* generally begin to grow by April (Croft and Preston, 2001), though the initiation of growth was observed to be much earlier in the Frome and by early April plants are generally 30 – 40 cm tall (Personal Observation). Biomass increases rapidly and the plants produce flowers from July to September and fruits may not ripen until November (Croft and Preston, 2001). *Sparganium erectum* begins to decline immediately after flowering and generally shows signs of obvious decline before *Ranunculus calcareous*. The plants generally decompose *in situ* because of the low velocities and, unlike *Ranunculus calcareous*, it is not usual for healthy plant parts to be transported downstream. Piquot *et al.* (1998) studied the reproduction strategies of *Sparganium erectum* in French rivers and found that asexual reproduction was favoured at the population level, i.e. at the scale of individual established stands, and plants spread underneath the substrate through rhizomes. However, because seed production is the only viable way of long distance dispersal (Piquot *et al.* 1998), seed production is favoured at the meta-population level, i.e. at the river scale. This discrepancy between species requirements at different scales is thought to partially explain the need for both sexual and asexual reproduction. *Sparganium erectum* flowers are both male and female and are usually pollinated by wind dispersal of pollen or through self-pollination.

APPENDIX B: MESOSCALE HYDRAULICS – DARCY-WEISBACH FRICTION FACTOR

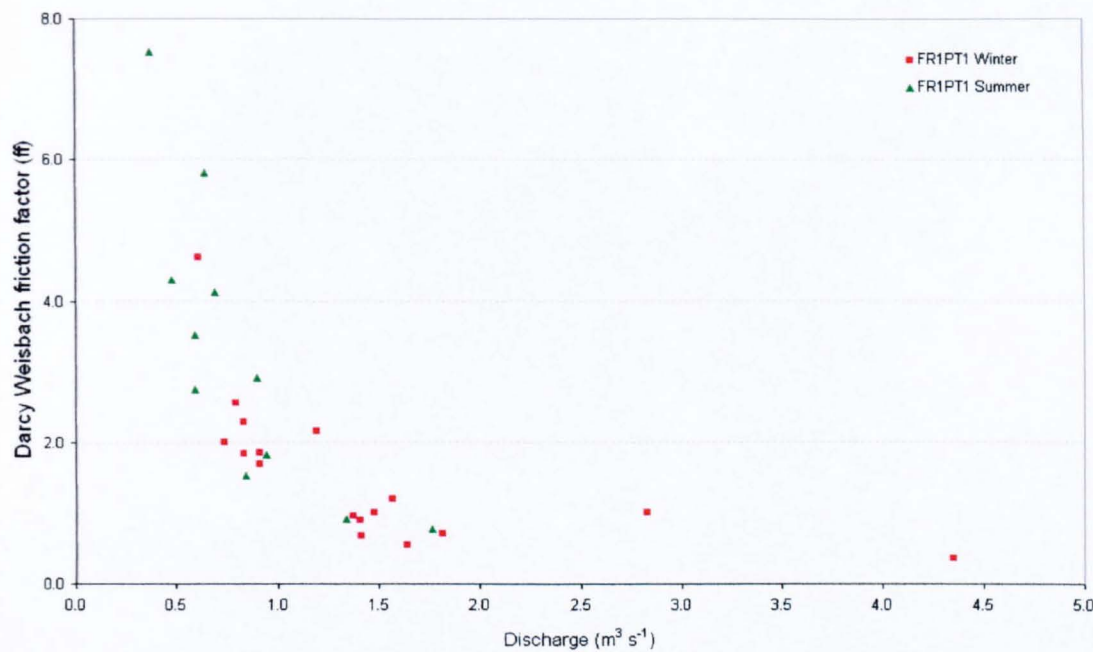


Figure 1 Darcy-Weisbach friction factor and discharge at Crockways (August 2003 – January 2005)

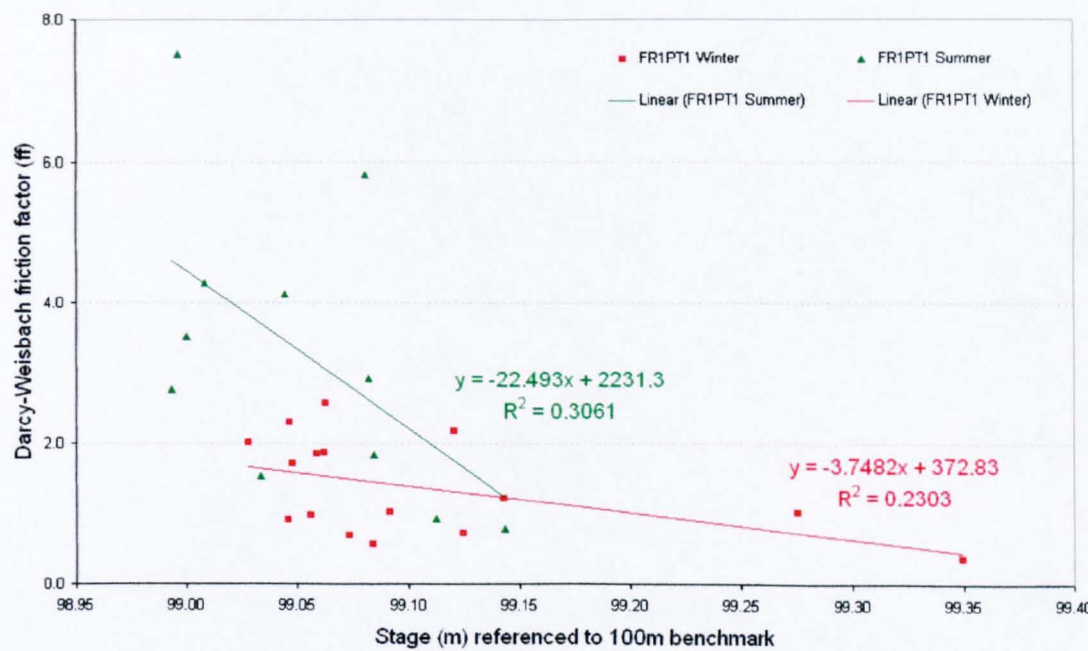


Figure 2 Darcy-Weisbach friction factor and stage at Crockways (August 2003 – January 2005)

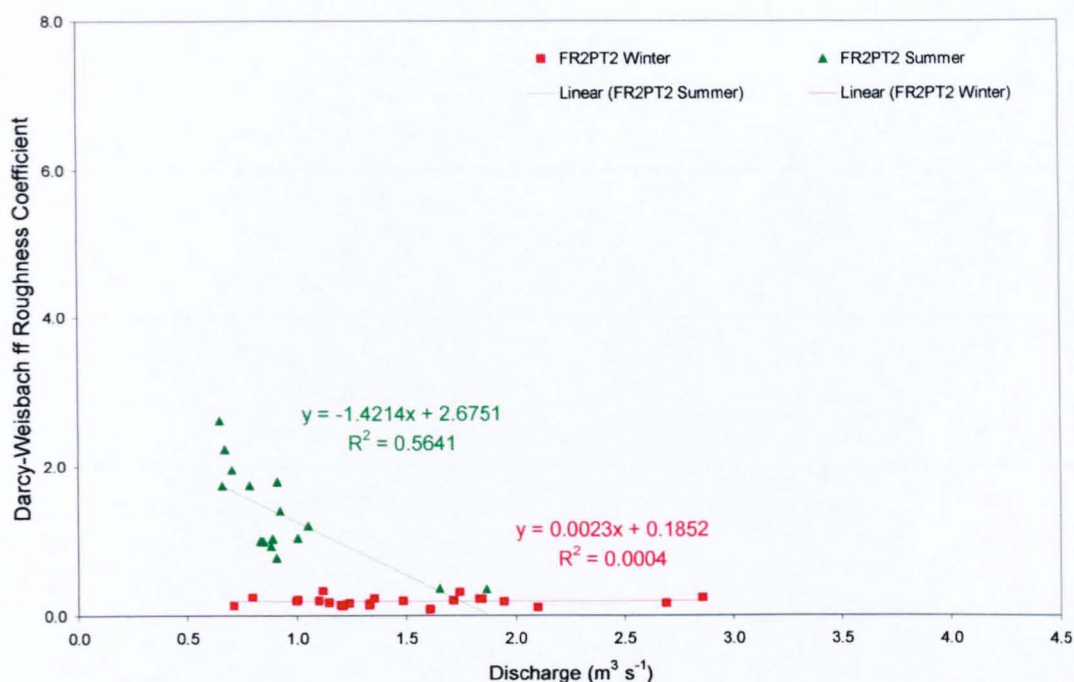


Figure 3 Darcy-Weisbach friction factor and discharge at Frampton (September 2003 – January 2005)

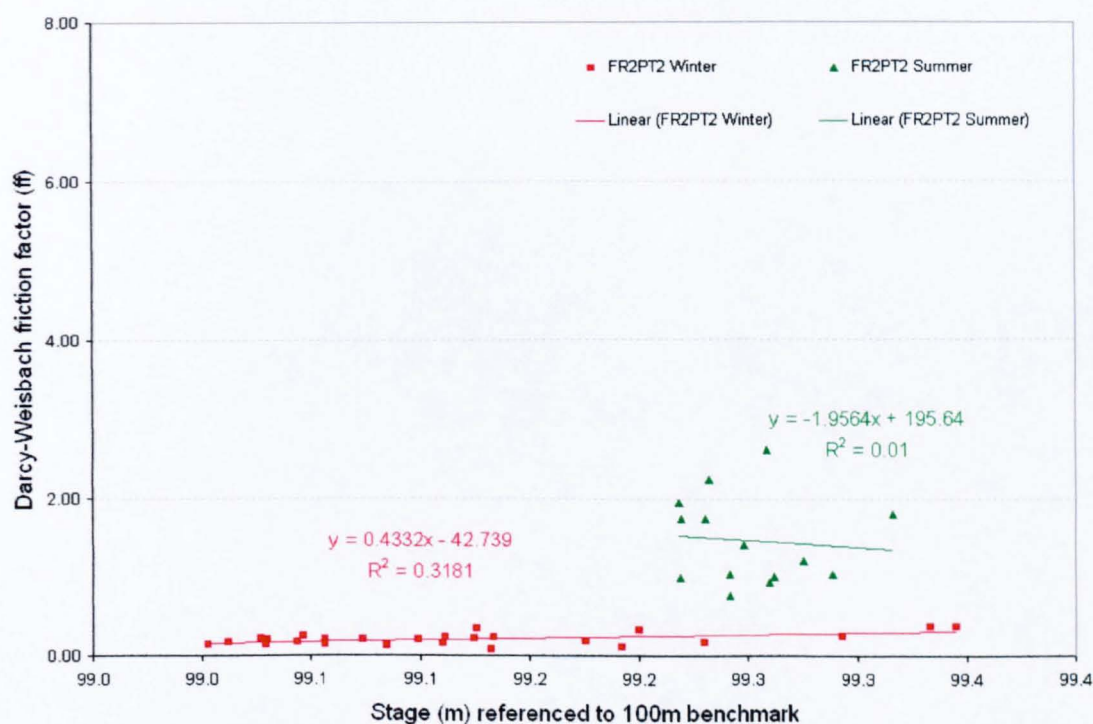
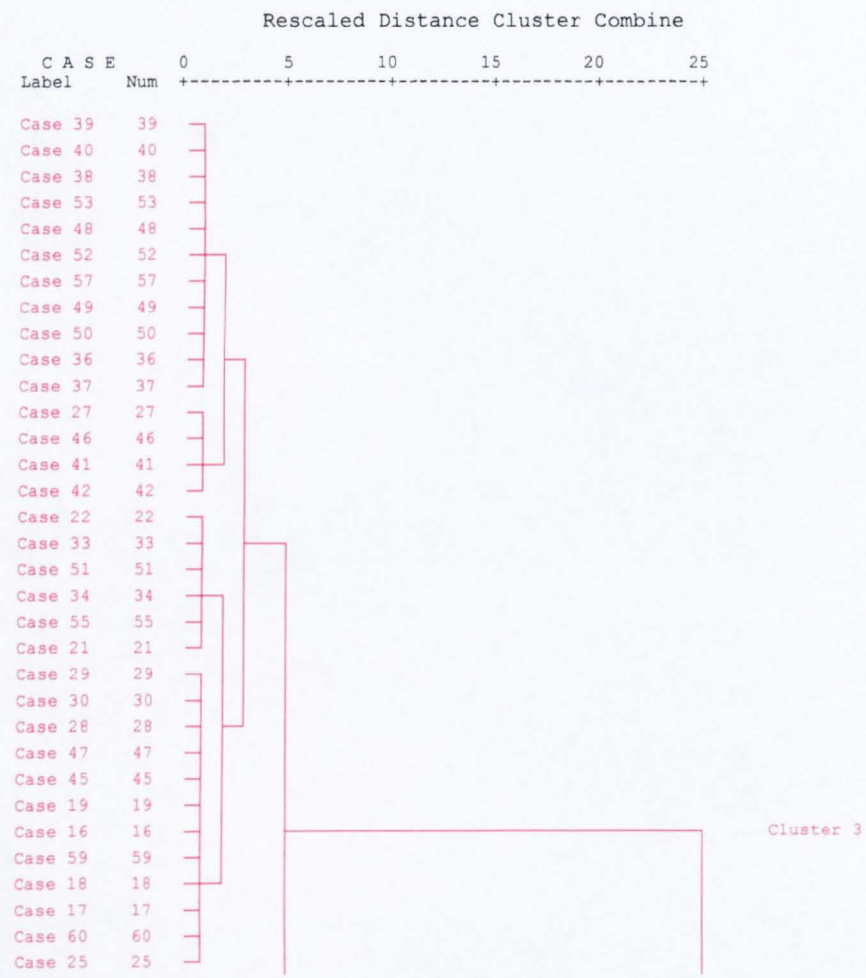


Figure 4 Darcy-Weisbach friction factor and discharge at Frampton (September 2003 – January 2005)

APPENDIX C: HEIRARCHICAL CLUSTER ANALYSIS – EXAMPLE DENDROGRAM



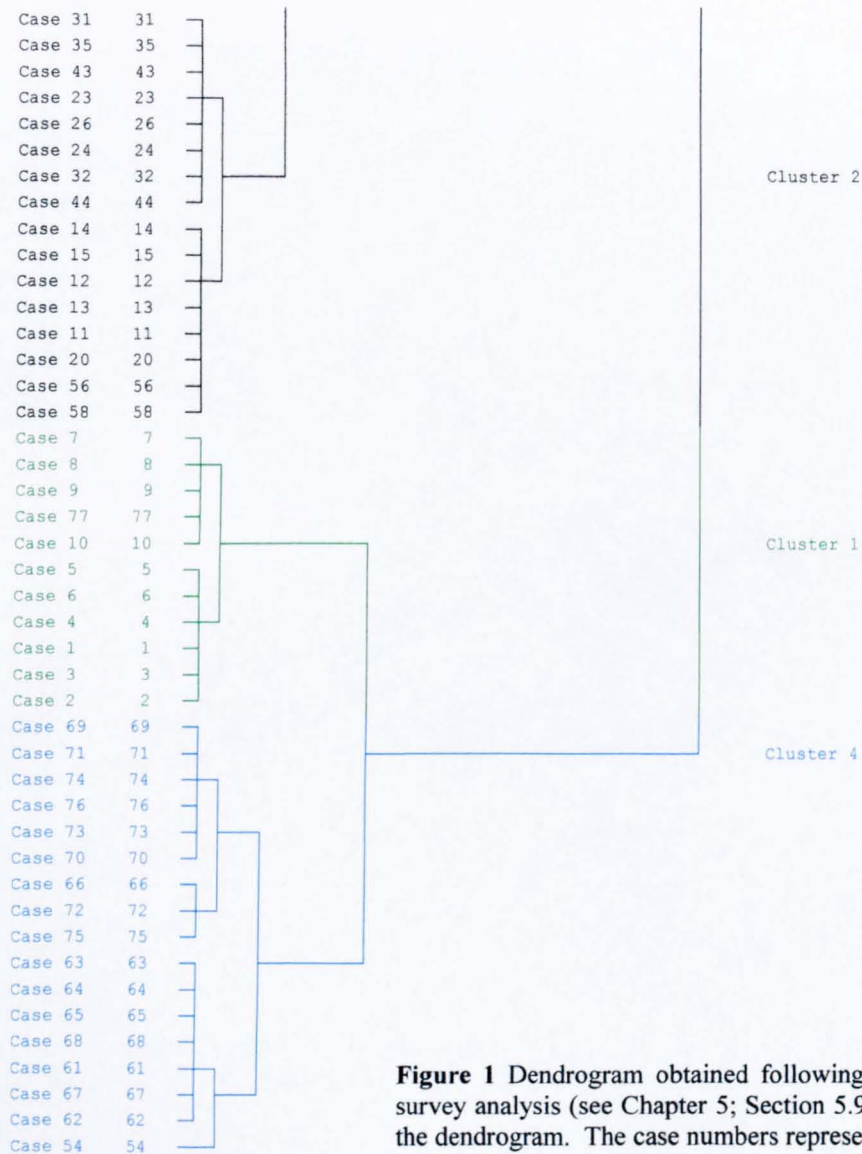


Figure 1 Dendrogram obtained following hierarchical cluster analysis using Ward's method for the Crockways five survey analysis (see Chapter 5; Section 5.9). The four clusters distinguished during the analysis have been indicated on the dendrogram. The case numbers represent individual velocity measurements at known spatial locations.

APPENDIX D GLOSSARY OF TERMS

ADV (Acoustic Doppler Velocimeter): equipment used to measure the speed of water flow using and adaptation of the Doppler principle.

Biomass: the total mass of a species or group of species under consideration.

Catchment: An area of land drained by a river and its tributaries.

Cross-section: A section formed by a plane cutting through a river channel at right angles to the direction of water flow (cf. transect)

Discharge: the volume of water passing a given river cross-section at a given point in time. Discharge is the product of cross-sectional area and velocity.

D₅₀: Median size of sediment particles in a sample or population; 50 %of the sediment particles are finer than the D₅₀ standard.

Emergent: self-supporting macrophytes which are rooted in under water sediment but the majority of their growth occurs above the water surface.

Groundwater: water stored in pores, cavities, cracks and other spaces in permeable and pervious rocks, usually bounded at its lower limit by an impermeable rock layer.

LOCAR (Lowland Catchment Research): a NERC (Natural Environment Research Council) thematic programme which aimed to facilitate detailed, interdisciplinary, research relating to the input-storage-discharge cycle and stream and wetland habitats of lowland groundwater dominated river systems.

Macrophyte: large aquatic plants visible to the naked eye. The term spans several taxonomic groups which grow in close proximity and respond to the same environmental constraints. Macrophytes can be classified according to their growth form and mode of attachment to the sediment and most classifications yield four major groups: (i) emergents; (ii) floating leaved and rooted in sediment (iii) submergent and (iv) free-floating.

Macroscale: research investigations or data collection which cover spatial ranges greater than 100m and within a timescale ranging from months to years.

Marginal: Macrophytic plants which grow in constantly wet soil or very shallow water and are generally found at the edge of rivers, ponds and lakes.

Mesoscale: research investigations or data collection at a spatial scale of 10s to 100s of metres and within a time framework of hours or days.

Microscale: research investigations at spatial scales of mm or cm and on a time framework of seconds or minutes.

PT (pressure transducer): equipment used to measure river stage by converting changes in pressure exerted by changing volumes of water into measurements of vertical distances following laboratory calibration.

***Ranunculus calcareous*:** submergent macrophyte species prevalent in chalk rivers.

Reach: a segment of a stream or river generally by distinguishing hydrologic features, e.g. from a confluence to a confluence, from one meander bend to the next or across a riffle-pool complex.

Reach-scale: research investigations or data collection undertaken across a reach.

Riparian: transitional zone at the interface between cultivable land and a river, populated by hydrophilic (water loving) plants.

RHS (River Habitat Survey)

River stage: the height of the water surface of a river above a fixed point.

***Sparganium erectum*:** emergent macrophyte species common on the River Frome.

Submergent – term which describes the growth form of macrophytic vegetation. Submergent refers to plants which grow predominantly beneath the water surface and derive some support from the surrounding water (cf. emergent)

Transect: A section formed by a plane cutting through a river channel, over a defined distance, parallel to the dominant flow direction, cf. cross-section

Turbidity: the degree to which light travelling through a water column is scattered by suspended particles, used as an analogue for sediment concentration.

Velocity: the speed or rapidity of water movement, measured in m s^{-1}

Water depth: the vertical distance between the water surface and the surface of the river bed.

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